The use of assisted succession to restore the globally imperiled dwarf pine-oak forests of the New Jersey Pinelands: An investigation of the functional and structural recovery of an abandoned gravel pit.

A Thesis

Submitted to the Faculty

of

Drexel University

by

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in partial fulfillment of the

requirements for the degree

of

Doctor of Philosophy

May 2010

Dedication

This work is dedicated to Joshua, Jordan, and Ayla Jiselle. My three amazing children who remind me daily never to take even the simplest of things in life for granted.

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Acknowledgements

My academic journey began many years ago and I never would have made it to where I am today without the support, friendship, and love of many people. First and foremost, the success of this doctoral work is a result of the dedication and commitment given to me by my supervising professor Dr. Walter Bien. I am so grateful for the relationship that has developed between the two of us over the years. I have been so fortunate to be trained and mentored by not only an excellent scientist but an extraordinary naturalist. Thank you Walt for sharing your passion for the Pine Barrens with me, I will always treasure our botanical explorations at Warren Grove Range. I am especially grateful that you held my work to the highest of standards and convinced me that my research was worthy of a Ph.D.

I am also thankful for the other brilliant scientists on my committee. To Dr. Kilham who was the first Professor to warmly welcome me to the department, and who always showed me support in whatever I did over the last 9 years. To Dr. Lacovara who taught me to separate emotion from science, but not the passion for science. To Dr. Dighton, who welcomed me into his Rutgers Pinelands Field Station, treated me as one of his own students, and showed me that what's below ground is as important as what's above ground. An especially heartfelt note of gratitude to Dr. Spotila, who understood and respected the difficulty in balancing family with academic commitments and gave me the freedom and space to conduct my own research, but was always there to see me through.

I would never have been able to accomplish the sophisticated soil analysis undertaken in this project without the guidance of and oversight by Dr. Dennis Gray. Dennis, I will always be indebted to you for all that you taught me and for dedicating so many hours helping me in the laboratory. I truly value your ongoing assistance and especially our friendship.

Many thanks to Dr. Hal Avery, who over the last nine years, always found ways to help me improve my presentations, my research questions, and my study design. My work at the Warren Grove Gunnery Range would not have been possible without the commitment and support of Captain Richard DeFeo, Dr. Doug Ripley and the staff and crew of the United States 177th Fighter Wing.

I could never have counted the many trees, measured the many blades of grass, or strung the many plots without the help and loyalty of my friends and peers at Drexel. Thank you, Ron S., Alicia B., Katherine B., Bryan W., Bryan F., Anika M., and Mark M.

I must also thank Ashley Olson who helped me to complete the final stages of this project by providing my three children with the best care, the greatest amount of love, and enough distraction to keep them out of my office while I wrote.

Finally, I would never have made it to where I am today without the support, devotion, motivation, friendship, and unconditional love of my beautiful and amazing wife Michelle, who continues to inspire me to live my dreams. Thank you for believing in me.

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Abstract

The use of assisted succession to restore the globally imperiled dwarf pine-oak forests of the New Jersey Pinelands: An investigation of the functional and structural recovery of an abandoned gravel pit

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Over the last 65 years, military operations at the New Jersey Air National Guard's Warren Grove Gunnery Range (WGGR) have scarred and fragmented pristine pine barren habitats within the Pinelands National Reserve. The Pinelands National Reserve in Central and Southern New Jersey contains one of the largest continuous areas of pine barren ecosystems in the world, including the globally imperiled dwarf pine-oak forests known as the pine plains. Because the pine plains are ecologically important and no specific recommendations for restoring pine plain habitats exist, I designed an investigation that would examine and evaluate past approaches to restoring pine plains communities, identify specific revegetation strategies that compliment natural processes, and experimentally determine the environmental constraints for reintroducing native communities. This investigation also quantified how highly disturbed pine plains system recover structurally and functionally from mechanical disturbance. Furthermore, this work will help to formulate an understanding of some of the ecological processes driving succession in these nutrient limited, fire adapted ecosystems.

I found that after 10 to 20 years of restoration, previous attempts to reforest degraded pine plains habitats produced monocultures of pines that were similar to reference plots in terms of total plant cover but were dissimilar in terms of plant community structure and composition. As a result of these data, an alternative approach

to reforestation was investigated. This approach used native colonizing grasses to ameliorate dysfunctional soil conditions, facilitate natural recovery and catalyze successional processes. An abandoned gravel pit restored with native grasses showed recovery patterns that were both structurally and functionally similar to those of a mechanically disturbed site recovering naturally by spontaneous succession. The establishment of native grasses during restoration also improved when soil amendments created ideal germinating conditions. This investigation demonstrated that using assisted succession to restore mechanically disturbed, pine plains habitats in the New Jersey Pinelands is superior to other previously applied reforestation methods.

CHAPTER 1: Introduction

BACKGROUND

Ecological degradation is a widespread global problem and is the leading cause of species extinction and ecosystem services decline (Ehrlich and Mooney 1983; Wilson 1988; Vitousek 1997; Dobson 2006; Worm 2006). As the human population continues to grow, degradation in the form of natural habitat destruction and over-exploitation of natural resources is almost certain to continue. In fact, the current rate of consumption due to population expansion may threaten our own stability as a species (Hobbs and Harris 2001). Although the earth is resilient and capable of self-repair, recovery from severe disturbances may simply require more time than is possible based on current demands. For a sustainable future, we must expand beyond preservation and conservation to include repair of habitats in our planning. Ecological restoration has quickly become one of the only viable options for rehabilitating damaged habitats, repairing ecosystem services and reducing the loss of biodiversity.

Ecological restoration has been described as the "recreation of entire communities of organisms, closely modeled on those occurring naturally" (Jordan et al. 1988). The Society for Ecological Restoration refined this definition to include any attempt to accelerate the recovery of a severely disturbed ecosystem unable return to its pre-disturbed condition without intervention (Society for Ecological Restoration 2002). In a more general sense, restoration encompasses any physical, chemical, or biological manipulation of a disturbed habitat into a desired state (Walker et al. 2007). Importantly,

true restoration emphasizes that an environment is returned to a self-sustaining ecosystem free of "continued intervention" (Berger 1993). Bradshaw (1983) referred to ecological restoration as an “acid test” for ecology in that our true understanding of an ecosystem arises from our ability to re-build it after studying the individual pieces. The instruction manual needed to reassemble the pieces develops from a synthesis of all ecological theories including community, population, landscape, and evolutionary ecology.

The amount of intervention necessary for restoration will depend on the extent of habitat degradation, the ecosystem in question, and the source and intensity of disturbance. Depending on the problem, restoration may require removing a perturbation or performing an engineered modification. The term ecological reclamation is often used interchangeably with ecological restoration. Generally, it is reserved for more technical interventions where the goal is not necessarily to restore a historic condition, but rather to recreate a socially acceptable alternative (Bradshaw and Chadwick 1980; SER 2000). Ecological reclamation is typically applied to the recovery of mined lands, but can be used to describe other types of restoration projects. Reclamation involves the return of derelict or denuded areas to some form of ecological productivity, but unlike restoration, ecological reclamation may not necessarily resemble pre-disturbed habitats. The objectives of reclamation may be limited to erosion control, hazardous waste reduction, aesthetic improvement, re-vegetation or return of some biologic productivity, while restoration goes further in establishing diversity, richness, and ecosystem function (Bradshaw and Chadwick 1980; SER 2000). In a sense, the process of recreating ecosystems can be part of a continuum between restoration and reclamation in which the

ideal goal is the establishment of functional integrity, sustainability, and continuity with the surrounding natural habitat.

Ecological restoration involves re-establishing the links between the pieces of an ecosystem; therefore two ecological concepts are critical for its practice. Successional processes and assembly rules have only recently become integral parts of the theoretical framework for restoration ecology (Templeton et al. 2004; Walker et al. 2007). Although fundamentally different, both address the dynamics driving community development, vital aspects of ecological restoration. An integration of these two concepts is necessary because restoration is a purposeful manipulation of succession which requires an understanding of how species assemble under varying environmental conditions (Lockwood 1997; Young 2000).

Determining restoration success can be problematic and the optimal approach has long been debated in the restoration literature (Hobbs and Norton 1996; Ruiz-Jaén and Aide 2005a). Pysek et al. (2001) have stated that a proper assessment of restoration technique is difficult without comparing the recoveries of both a restored site and natural site developing without human intervention. Because of the difficulty in finding suitable naturally recovering sites, restoration success is often determined by comparing restored sites with historic references or a target system, but this has recently been brought into question in the face of global climate change (Harris et al. 2006; Millar et.al 2007). Restoration ecologists do agree that success will ultimately depend on the project goals. For example, if the aim is to recreate a sustainable ecosystem from a dysfunctional one, long term monitoring is required. On the other hand, if the goal is to minimize erosion and sedimentation into a watershed, a shorter commitment may be appropriate. In most

restoration cases, financial resources prevent extended management beyond the restoration act itself. True restoration, the development of a functional self sustaining ecosystem resembling a former pre-disturbed community, is not immediate and can take many years, if not hundreds of years, before success is achieved.

Because of the difficulty in determining true restoration success, it is imperative to get it right the first time. This requires finding a strategy that not only minimizes the amount of restoration required, but relies on nature's innate resilience and self-sustainability (Bradshaw 1997). A promising approach to restoration is to manipulate or catalyze spontaneous succession. Many sites do not undergo succession naturally because soil conditions are dysfunctional or sites are biologically inactive. This is especially true in landscapes disturbed by surface mining. By finding techniques to ameliorate these problems, dysfunctional habitats may be restored to a threshold at which point succession can proceed naturally (King and Hobbs 2004; Prach et al. 2007). Assisted succession represents an intermediate approach to restoration. The minimal approach is to do nothing and hope succession occurs spontaneously. However, this is often unsuccessful because natural processes are so damaged that natural recovery is impossible. Additionally, leaving sites alone may allow them to become vulnerable to invasives which may alter desirable successional trajectories. At the other end of the gradient is a more intensive or technical restoration. Technical restorations attempt to direct or accelerate succession towards desirable targets by extensively manipulating biotic and abiotic factors. In many cases, target habitats are not sustained because important successional steps were bypassed (Higgs 1997). The ideal approach to return a derelict site to some level of productivity depends on many factors, but identifying the minimal

amount of intervention necessary to repair successional trajectories is a general aim for ecological restoration (Bradshaw 2002).

THESIS OBJECTIVE AND RATIONALE

Over the last 65 years, military operations at the New Jersey Air National Guard's Warren Grove Gunnery Range (WGGR) have scarred and fragmented pristine pine barren habitats within the Pinelands National Reserve. The Pinelands National Reserve in Central and Southern New Jersey contains one of the largest continuous areas of pine barren ecosystems in the world, including the globally imperiled dwarf pine-oak forests known as the pine plains. Because the pine plains are ecologically important and no specific recommendations for restoring pine plain habitats exist, I designed an investigation that would examine and evaluate past approaches to restoring pine plains communities, identify specific revegetation strategies that compliment natural processes, and experimentally determine the environmental constraints for reintroducing native communities. This investigation also quantified how highly disturbed pine plains system recover structurally and functionally from mechanical disturbance. Furthermore, this work will help to formulate an understanding of some of the ecological processes driving succession in these nutrient limited, fire adapted ecosystems.

Specifically, this investigation addresses the following questions:

1) Are previously tested strategies for reclaiming highly disturbed landscapes on the Warren Grove Range adequate for restoring the ecological integrity of pine plains habitat?
2) Is there a more effective ecological approach which provides advantages over previously tested restoration strategies?

3) What are some of the ecological factors that limit the success of restoration in the Pine Barrens?
4) Can successful methods for catalyzing succession on dysfunctional pine barren soils be developed?
5) Are similarities in aboveground structure complimented by similarities in belowground function when comparing a naturally recovered site with a restored site?

ORGANIZATION OF CHAPTERS

Chapter 2: An evaluation of reforestation on highly disturbed sites in the New Jersey Pinelands, USA

The purpose of this study was to compare restoration outcomes among multiple restoration projects undertaken over the last 20 years in the pine plains community at WGGR. I indentified 8 of 15 restored sites at WGGR that were either reforested by planting pitch pine seedlings or seeded with pitch pine and shrub oaks. All sites were mechanically disturbed and were either former targeting areas that were surface plowed or gravel pits that were excavated. I compared plant community structure and species composition among the eight sites with an undisturbed pine plains reference site to best identify the restoration strategy that produced a plant community most similar to the surrounding pine plains community. This study evaluated the efficacy of seeding or planting target species (pines and oaks) in restoring mechanically disturbed pine plains communities.

This study tested the following hypotheses:

H_0: The plant community structure of restored pine plain landscapes do not differ from surrounding "undisturbed" landscapes.

H_a: The plant community structure in different restoration sites will vary as a result of the revegetation strategy utilized and disturbance history.

Chapter 3: A Case Study of Assisted Succession for the Restoration of an Abandoned Gravel Pit in the New Jersey Pinelands

The purpose of this study was to evaluate the use of assisted succession as a method for restoring mined or cleared landscapes in upland forests of the New Jersey Pinelands. A warm season native grass (*Schizachyrium scoparium*) was seeded with soil amendments on a graded portion of an abandoned gravel pit. Two years after seeding, the site showed signs of community recovery similar to those occurring on a naturally recovering mechanically disturbed site in the pine plains. In this study, I compared plant growth, cover and composition of a restored gravel pit with a 20 year old site recovering by spontaneous succession.

This study tested the following hypotheses:

H_0: The establishment of *S. scoparium* on a reclaimed gravel bed in the New Jersey Pine Barrens does not differ in structure (density, cover, growth) from a disturbed site undergoing spontaneous succession.

H_a: Successional trajectories between a natural recovering site differ from those of a site restored using assisted succession.

Chapter 4: Factors contributing to successful establishment of *Schizachyrium scoparium* on an abandoned gravel pit

Schizachyrium scoparium, a native, warm season, perennial grass, has been shown to be a good pioneer species for severely disturbed upland landscapes. In this chapter, I present the results of a germination field trial that sought to identify factors that contribute to successful germination and establishment of *S. scoparium* on abandoned gravel spoils in the New Jersey Pine Barrens. This experimental study examined the effects of mulch, fertilization, and tillage on germination and establishment of *S. scoparium* on eroded soils. Furthermore, this study helped to develop a restoration model using *S. scoparium* for abandoned gravel pits and other mechanically-disturbed upland Pine Barren habitats while providing recommendations for reclaiming and managing disturbed sites for natural resource managers at WGGR and the New Jersey Pinelands.

This study tested the following hypotheses:

$H_{o:}$ Germination (number of emerged seedlings) and subsequent establishment (growth and survival) of *Schizachyrium scoparium* on a reclaimed gravel bed in the New Jersey Pine Barrens is not influenced by seedbed preparation, which includes the addition of organic amendments, the application of nutrients, tillage, or any combination thereof.

H_{o1}: Differences in community structure (density, cover, growth) are not related to differing thicknesses of mulch coverage

H_{o2}: Differing thicknesses of mulch coverage and nutrient application do not influence germination and establishment.

H_{o3}: Germination is unaffected by tilling and incorporating organic material into the soil

H_a: Germination and establishment of *Schizachyrium scoparium* on a reclaimed gravel bed in the New Jersey Pine Barrens is a function of the type of revegetation technique (mulching/tillage/fertilization) utilized.

Chapter 5: A Comparative Analysis of Belowground Function between a Restored Gravel Pit and Naturally Recovering Disturbed Pine Plains Community in the New Jersey Pine Barrens

The purpose of this chapter was to ascertain whether similarities in above ground plant structure between a restored site and naturally recovering site reflect similarities in belowground function in terms of nutrient cycling, nitrogen allocation, mycorrhizal associations, and root development. By comparing belowground function between the two sites, I determined if using native warm season grasses helped catalyze successional processes in disturbed sites that showed no evidence of recovering on their own. This study also provided information on the successional dynamics of disturbed pine plain and demonstrated how these systems recover from disturbance naturally or by restoration.

The following hypotheses were tested in this chapter:

$H_0$1: On mechanically disturbed sites in the pine plains, there are no differences in nitrogen allocation and nitrogen availability between sites restored by assisted succession or sites recovering by spontaneous succession.

$H_0$2: Because of similarities in above ground structure between these two sites, there are no differences in root development patterns.

$H_0$3: The amounts of root length colonized by mycorrhizae between plants of a restored site are no different than those of a naturally recovering site.

H_a: Similarities in aboveground structure between a restored and naturally recovering site may be driven by different functional processes as measured from belowground.

CHAPTER 2: An evaluation of reforestation on highly disturbed sites in the New Jersey Pinelands, USA

ABSTRACT

Within the last few decades, military training areas have been recognized worldwide as reserves for native flora and fauna. It follows that these protected areas serve as important refuges for rare or endangered species. With one of the largest globally imperiled dwarf pine plains communities, the Warren Grove Gunnery Range in south central New Jersey serves as such an important refuge. Over the last 66 years, military operations degraded portions of these upland pine barren habitats prompting restorative efforts. Although some general guidelines for reclaiming native plant communities exist, there still are no specific recommendations for restoring these landscapes. In order to identify reforestation strategies that are most effective, I performed a comprehensive ecological evaluation of both plant community structure and species composition in eight restored sites of differing ages and design. I found that after 10 to 20 years of restoration, total plant cover returned to the oldest sites but no revegetation strategy completely replaced the plant community composition or structure observed in a reference system. Species abundances were variable among all sites with pine tree density ranging from 2.0 trees/m^2 in seeded sites to 0.28 trees/m^2 in planted sites. Although species richness generally increased with time, all restoration sites lacked the shrub and oak densities observed in reference plots. These data suggest that on short temporal scales, current restoration strategies are effective in replacing vegetative cover but ineffective in restoring a natural community structure.

INTRODUCTION

Worldwide, military training lands have been identified as areas having high biodiversity (Gazenbeek 2005, Warren et al. 2007). More so, military installations have been recognized as refugia for endangered or threatened flora and fauna (Goodman 1996; Boice 1997; Warren and Büttner 2007) with lands managed by the US Department of Defense (DoD) accounting for a greater number of rare species per unit area than lands maintained by any other US Federal Agency (Cohn 1996). With approximately 11 million acres of DoD land widely distributed throughout the United States, it's not surprising military training areas harbor a variety of rare native landscapes and species. However, land area alone does not account for high biodiversity. Other reasons exist for the disproportionate number of rare species observed on military installations. First, large tracts of military land remain undeveloped and publically restricted. These areas serve as important safety buffers between active training sites and surrounding developed areas. For example, at the US Army's Yuma Proving Ground in Arizona only 10% of the installations 350,000 hectares are used for military readiness leaving the remaining land natural and protected (Cohn 1996). Secondly, military activities can produce heterogeneous landscapes which increase diversity and may even be favored by rare disturbance dependent species (Drigot 2001; Smith et al. 2002; Leis et al. 2005; Warren and Büttner 2007; Bien et al. 2009). The combination of protected intact areas and highly disturbed trafficked areas that characterize military training lands create a mosaic of landscapes which can support habitats rich in biodiversity (Warren et al. 2007).

Although military training is crucial to assure military readiness, the DoD is not exempt from laws that protect rare or threatened species and in fact is directed to do so.

Military installations are mandated to develop and comply with an Integrated Natural Resources Management Plan (INRMP) that provides for environmental stewardship and oversight (Ripley and Leslie 1997). Given the nature of military training, disturbance to sensitive landscapes can often be widespread and, in some ecosystems, destructive. Land managers require practical tools to assist them with the recovery of these disturbed areas, thus restoring the biological productivity to pre-disturbed levels. Ecological restoration, the process of closely modeling entire communities of organisms after those occurring naturally, has become such a tool (Jordan et al. 1988). This process attempts to hasten the recovery of severely degraded landscapes that, without intervention, cannot recover on their own (Society for Ecological Restoration 2002). Restoration allows land managers to assist or simulate the natural recovery of disturbed landscapes returning them on successional trajectories that are similar, if not equivalent, to surrounding natural plant communities (Bradshaw 1996). Thus, with successful restoration, the DoD can promote the sustainability of the same natural landscapes that the military depends on for their training. At the Air National Guard's Warren Grove Gunnery Range in New Jersey USA, ecological restoration has been an important part of the INRMP (New Jersey Air National Guard 2001).

The Warren Grove Gunnery Range (WGR) occupies approximately 3,801 ha of the 450,000 ha within the East Pine Plains region of the Pinelands National Reserve (PNR). The PNR, designated as the United States first National Reserve, is one of the largest contiguous pine barren ecosystems in the world (McCormick and Forman 1998). Upland pine barren habitats are characterized by well-drained, sandy, nutrient-poor soils, with a high frequency of fire that results in a unique plant community of fire adapted

species (Lutz 1934; Good and Good 1975; Buchholz and Good 1982). A gentle rolling topography influences a mosaic pattern of upland and lowland habitats with high ecological value (Good et al. 1998). The PNR supports approximately 9,000 ha of dwarf pine plain forest type, a unique and rare ecosystem (Harshberger 1916). With a range restricted to coastal plain portions of New York and New Jersey, and pockets of acidic bedrock outcroppings in Massachusetts and West Virginia, the dwarf pine plain forests are considered a globally imperiled habitat type (Natureserve 1998). The dwarf pine plains are characterized by a multi-stemmed diminutive form of pitch pine (<3 meters) with a sprawling growth habit. This diminutive stature, along with serotinous cones, was likely influenced by historically high fire frequencies (Good et al. 1998).

Over the last 66 years, military operations at WGR seriously degraded significant portions of dwarf pine plains habitat. Military operations that caused landscape disturbance include air to ground weapons delivery, road maintenance, soil excavation, installation of fire breaks, and prescribed burning. These operations disturbed approximately 162 ha of pine barren forest types at WGR (Windisch, unpublished data). Disturbed or denuded habitats destroy the ecological integrity of the pine barrens by causing habitat fragmentation, erosion, sedimentation and unnatural firebreaks (New Jersey Pinelands Commission 1980; Gibson et al. 1988).

Although several studies examined ways to restore upland pine barren habitats, most have focused on the effects of long term fire suppression and not mechanical disturbances (e.g. soil extraction) (Radeloff et al. 2000; Nielsen et al. 2003). In the absence of frequent forest fires, hardwoods (e.g. white oak) become the dominant tree type, eventually replacing pines. Selective thinning of hardwoods and prescribed burning have been the

most common restorative practice applied for reversing the effects of long term fire suppression (Radeloff et al. 2000; Rhoades et al. 2002). However, soil extraction or land clearing practices in the pine barrens have resulted in either the partial or complete removal of upper soil horizons. These cleared oligotrophic areas don't recover quickly and often remain depauperate for decades (Bradshaw 1997). Research on restoring cleared or mined upland pine barren landscapes has been limited.

Restoration first became a priority for land managers at WGR in 1988. Since then there have been 15 restoration projects, all located within the active training area of the installation. In 1989, studies were conducted to identify the best methods for restoring mechanically disturbed sites. Researchers investigated seedling establishment (Fimbel 1992) and trials in density planting and soil amendments. For example, Fimbel and Kuser (1993) demonstrated that one successful method for restoring dwarf pitch pine forest types was to augment planting of pitch pine seedlings with 16 Mg of compost sludge per hectare. However, the New Jersey Pinelands Commission restricted the use of composted sewage in subsequent restoration projects due to the potential for heavy metal contamination and excess nitrogen loading in ground water. As a result of these studies, several general guidelines were developed for restoring native plant communities but few specific recommendations were made for restoring natural system dynamics. In order to ascertain the ultimate efficacy of any restoration project, long term studies are imperative. Without such evaluations, determining restoration success and interpreting its conceptual relevance to ecology overall is limited (Hobbs and Norton 1996). To date, there have been no published long term studies to assess the recovery of these restored areas at WGR. The purpose of this study was to conduct a comparative analysis of restoration

projects in varying stages of recovery in dwarf pine plains communities at WGR. Specifically I compared plant community structure and species composition among eight restored sites with an undisturbed pine plains reference site to best identify the restoration strategy that produced a plant community most similar to the surrounding pine plains community. By identifying the most effective restoration strategies, land managers will be better able to restore disturbed pine plains habitats.

METHODS

1) Study Area and Site Selection

I conducted this study at the Warren Grove Gunnery Range in Burlington County, New Jersey (lat 39°41'N, long 74°23'W) (Appendix 1). Regional annual temperature ranged 0 to 24°C with annual rainfall of 1100 mm. The area was located on the Outer Atlantic Coastal Plain physiographic province where upland areas were characterized by dry, sandy, oligotrophic, and acidic soils (Woodmansie-Lakehurst Association). Dwarf pitch pine (*Pinus rigida*) was the dominant tree along with shrub oaks, *Quercus marilandica* and *Quecus ilicifolia*. Arborescent pine and oak trees were notably absent. The shrub layer was dominated by ericaceous species such as *Gaylussacia baccata, Gaylussacia frondosa, Kalmia latifolia, Vaccinium palidum* and the New Jersey state endangered *Corema conradii*. Ground cover was primarily comprised of lichens, mosses, grasses (*Andropogon virginicus*, *Schizachyrium scoparium*), sedges (*Carex pennsylvanica*, *Carex tonsa*), forbs (*Melampyrum lineare*) and sub-shrubs (*Arctostaphylos uva-ursi*).

I used aerial photos, ground surveys, published and unpublished reports to identify 15 unique restoration projects within the training areas of WGR. These restored sites typically had one of two prior land uses. Sites were either former target or training areas that were cleared by bulldozing or abandoned gravel pits. Restoration at WGR first began in 1988 and ended in 2001. Several revegetation strategies were employed in each project. Sites were either planted only with nursery grown pine seedlings, planted with nursery grown pine and oak seedlings, or seeded with pines and oaks. In addition, some sites were planted with the switchgrass, *Panicum virgatum*. No more than three plant species were seeded or planted at one time and planting densities varied among sites. Site preparation also differed from site to site with some sites undergoing no preparation while others were tilled and amended with fertilizer. Because I was interested in identifying revegetation strategies that were most effective in restoring plant communities similar to undisturbed dwarf pine forests, I excluded sites in which I was unable to clearly identify a systematic or reproducible methodology. Because sites differed in size, I only examined sites that had restored areas greater that 0.2 ha. Smaller sites were excluded because of the potential influence of edge effect on my data. In total, my study compared 8 of 15 restoration projects. See Table 1 for a summary of land use histories and revegetation strategies employed at each site and Figure 1 for the location of each site.

2) Sampling:

To compare differences in community structure and species composition among sites and between a reference site, I randomly installed between two and eleven 5 x10-m plots in each unique restoration project using a coordinate system specific for the size and shape of each. The number of sample plots varied for each site due to the heterogeneity

of size, shape, or structure of the restored area. As such, less sample plots were installed in smaller restoration sites while a greater number of sample plots were installed in larger sites or non-contiguous sites which contained roads, firebreaks, or patches of intact forests. Plot size was based on the maximum sampling area which could fit in narrow sites. A total of 30 plots were installed across the eight restoration sites. Six 5 x10-m plots were installed in a reference forest representing intact pine plains habitat. The reference system was located approximately 1 to 2 km from restoration sites and was last burned by wildfire in 1959. I compared pooled plot data for each restoration site with pooled data from the six reference plots.

I assessed plant community structure by measuring cover and density of all vascular plants. I also measured height, foliage cover and basal area for all trees. I assessed species composition by inventorying all plant species within each plot at the beginning and end of the growing season. Although the importance of non-vascular plants has been reported for soil development in pine barren habitat (Sedia and Ehrenfeld 2003) only vascular plants were inventoried in this study. Plant identification followed the nomenclature of Kartesz (1999). I used the line intercept method to measure percent ground cover for each plant species (Bullock 1996) and for determining relative frequency of each species (Brower et al. 1998). I calculated plant densities by counting all trees and shrubs within each 50-m^2 plot. I nested two 1x1-m subplots in opposing corners of each 50-m^2 plot to measure forb density. Tree basal area was calculated from stem diameters at ground height (root collar) since mature pygmy pines may not reach the standard diameter measure at breast height (Westman and Whittaker 1975). Because dwarf pine trees often have an asymmetrical growth habit, I measured the short and long

axis of each tree crown and averaged the two measures to calculate crown diameter and foliage cover (adapted from Brower et al. 1998). Values of foliage cover greater that 100% were possible due to canopy overlap. Tree size class was also evaluated by identifying the number of trees below 1-m, between 1 and 2-m, and greater than 2-m. Sampling took place during the growing seasons between April 2003 and July 2005.

3) Data analyses

In order to ascertain which revegetation strategies were most effective, I used both univariate and multivariate data analysis. Restored sites with non-contiguous areas were first analyzed individually to determine if each separate area required separate treatment. I found no differences between the separated areas of unique restoration sites and pooled all plots in these sites. For example, site 1988 consisted of 6 separate target site lines restored simultaneously. There were no significant differences among these sites when compared as individual treatments. Therefore, these sites were pooled as a single restoration project for subsequent analysis.

I used Simpson's index as the measure of diversity. This index took into consideration the number of species present, the total numbers of individuals in a sample, and the proportion of the total that occurred for each species measured (Brower et al. 1998). I used a one way analysis of variance (ANOVA) to compare plant structure among sites. Assumptions of normality and heterogeneity of variance were tested using the Kolmogoriv-Smirnov Test and the Levene Statistic respectively. Square root or arcsine transformations were performed where data exhibited non-normality. I accepted statistical significance at $p = 0.05$. I used the Fisher's protected Least Significant Difference (LSD) post-hoc test to examine differences among sites. Alternatively, I used

the Kruskal-Wallace and Mann-Whitney U tests to identify differences among group means when parametric assumptions could not be met. Statistics were performed using the SPSS Statistical Software package, Version 15.0 (SPSS 2001)

I used cluster analysis and non-metric multidimensional scaling (NMDS) to spatially identify similarities in plant community structure and species composition between restored sites and reference sites. The data matrix for the cluster analysis consisted of nine sites (eight restored and one reference) and 14 structural characteristics. I used a hierarchical approach to clustering using nearest neighbor pair-group fusion strategies with Bray-Curtis distance coefficients (McCune and Grace 1995). Although no explicit assumptions were required of data used for cluster analysis, I log transformed structural data to minimize the bias of variables with larger variances (McGarigal et al. 2000). I visualized compositional differences in species abundances between sites in an NMDS ordination which spatially organized plots into n-dimensional space that minimized inter-plot Bray-Curtis distance measurements (McCune and Grace 2002).

To evaluate the significance of separation visualized in the ordinations, I ran multi-response permutation procedures (MRPP) operating on Bray-Curtis distances for the plot by abundance data matrix (Zimmerman et al. 1985). I reported results in terms of the generated A or "Agreement" statistic which served as a chance-corrected indicator of within group similarity. Where A = 1 groups are considered identical and when A = 0, groups are different. Values below 0.1 are common in community ecology (McCune and Grace 2002). To minimize the Type 1 error rate for MRPP pair wise comparisons, I applied the Bonferroni approximation to significance testing which resulted in a decreased α level of significance (adjusted $\alpha = 1/C\ [\alpha]$, where C= the number of pair wise

tests) (Abdi 2007). Cluster and ordination analyses were performed using the PC-ORD software package (McCune & Mefford 1995).

RESULTS

1) Tree Densities

Pine densities ranged from 13.9 to 102 trees /50 m^2 and were significantly different between sites ($F_{[8,27]} = 19.3$, $p < 0.001$) (Table 2). Only 4 of the 8 restored sites had pine tree densities similar to the reference site. Pine tree densities in two of the sites (1996B and 1996C) were three times greater than densities observed in the reference site. The reference site had significantly higher oak tree densities than any of the 8 restoration sites (Kruskal-Wallace $p < 0.01$). Oak densities in restoration sites were significantly less than the reference site but there were no differences among the restored sites.

2) *Cover*

Pine tree ground cover was significantly different among sites ($F_{[8,26]} = 2.5$, $p < 0.05$). Only the most recently restored site (1997) had significantly less pine cover than the reference site. There were significant differences in pine tree foliage cover ($F = 7.6_{[8, 26]}$, $p<0.01$) with cover greatest in site 1993A (106%) and lowest in 1996B (38%). Site 1993A was the only site with similar cover to the reference site (Table 2). The reference site had the highest pine basal area but was only significantly greater than three of the restored sites ($F = 4.5_{[8, 26]}$, $p < 0.01$).

Oak ground, foliage, and basal area the restored sites were significantly less than the reference site (Kruskal-Wallace $p = 0.001$, $p = 0.001$, $p = 0.001$ respectively). Oaks were

completely absent in two of the eight restoration sites. The restoration site had almost 10 times more oak foliage cover than the six restored sites with oak trees (Table 2).

Total vegetative cover differed among sites ($F = 2.796_{[8,25]}$, $p < 0.05$). Sites 1996B and 1997 had significantly less total vegetative cover when compared to all other sites. Shrub cover in all restored sites was significantly less than shrub cover observed in the reference site (Kruskal-Wallace $p < 0.001$). Site 1993B had greater forb cover (Kruskal-Wallace $p < 0.001$) than other sites including the reference site.

3) Tree Height

The tallest pine trees occurred in sites 1988, 1993A, and the reference site (Kruskal-Wallace $p < 0.001$) (Table 2). Site 1996B had the highest mean number of pines (per 50 m^2) in the smallest class (<1 m) while both the reference site and site 1993A had the greatest number of pines in the largest size class (>2 m) (Figure 2). Oak trees in the reference site were significantly taller than all restoration sites (Kruskal-Wallace $p < 0.001$). Oak tree heights among restored sites with oaks present were not statistically different from one another.

4) Species Composition

A total of 22 vascular plant species belonging to 12 families were inventoried throughout the sites (Table 3). Species richness varied among restored sites and ranged from 3 to 16 species. The reference site had the greatest number of species (n = 13). Although *Panicum virgatum* was used for restoration plantings in sites 1988 and 1993A, it was absent during the inventory.

Sites 1993C, 1996A, 1996B and 1997 were essentially monocultures of pitch pine (*Pinus rigida*) (Figure 3). In restoration sites, *P. rigida* comprised at least 73% of the total

number of individual plants compared to 20% for the reference site making it the most dominant tree species among all restored sites. The most frequent forb in restoration sites was *S. scoparium* (5 to 26%) while the forb *Melampyrum lineare* was most frequent in reference sites (2.5%). The shrubs *G. baccata, G. frondosa* and *V. pallidum* represented the greatest number of individuals (71%) in the reference sites. In general, shrubs were absent in all restoration sites except site 1988 where *V. pallidum* was most frequent.

The cluster analysis of all measured structural attributes (Listed in Table 2) resulted in separation of sites by age (Figure 4). Sites 1996B and 1996C were the most structurally similar, followed by sites 1993B and 1993C. These latter two sites formed a cluster which separated out at a greater distance than did older sites. The reference site separated out at the farthest distance indicating that it was the most structurally dissimilar.

The spatial ordination of species abundance by Non-Metric Dimensional Scaling Ordination (NMDS) showed a separation of reference sites from the restored sites and a weak separation of site 1988. There appeared to be no clear distinctions among remaining sites (Figure 5). The final solution had two dimensions (Stress = 8.72, $p = 0.02$) with both axes accounting for 93% of the variation. Subsequent multi-response permutation procedures supported the observed spatial separation in compositional structure between restored and reference sites ($T = -8.62$, $A = 0.4$, $p < 0.001$). Pair wise comparisons evaluated using MRPP showed the reference site and oldest site (1988) to be most dissimilar in terms of species abundance (Table 4).

DISCUSSION

This study compared the vegetation structure and composition of eight restored pine plains sites to adjacent areas representative of dwarf pine forests in the New Jersey Pinelands. Past ecological restoration of highly disturbed sites using different revegetation strategies produced varying results suggesting that restoring severely disturbed sites in these upland pine barren communities can be challenging. Although some restoration strategies produced a similar vegetative cover when compared to a natural reference site, no single strategy replicated plant community structure and species composition. The lack of differences in vegetative cover between restoration sites and reference plots was strongly influenced by the dominance of pitch pine in the restoration sites. Pitch pine accounted for 95% of the total vegetative cover in restoration sites compared to 73% in the reference site even though total vegetative cover did not differ. Restoration sites were most dissimilar to the reference site in terms of oak and shrub density and cover. Although some restoration sites included oak plantings or seeding, no site included a strategy for establishing shrub cover.

Without a specific revegetation strategy for restoring native shrub communities, recovery and recruitment of native blueberry (*Vaccinium* sp.) and huckleberry (*Gaylussacia* sp.) will be delayed. Little and Moore (1949) found that old field pine stands were invaded by hardwood trees long before the establishment of any shrub species. In abandoned pine plantations within the New Jersey Pinelands, assemblages of low bush blueberry and huckleberry were sparse (Laycock 1967). These data suggest that recruitment of shrub species is limited and may be driven by seed dispersal, rhizome development, and availability of safe microsites with adequate soil conditions (Eriksson

and Ehrlén 1992; Mejias et al. 2002). Furthermore, highly disturbed soils may be depleted of essential mycorrhizal symbionts limiting shrub establishment (St. John 1990; Hutton et al. 1997). Limited recruitment success in harsh soil conditions indicates that seed propagation of clonal ericaceous shrubs is difficult (Eriksson 1993; Eriksson and Fröborg 1996; Cairney and Meharg 2003) especially in pine barren habitats (Matlack et al. 1993). My results showing the lack of shrub development 20 years after restoration suggest the need to develop a more effective revegetation strategy for reintroducing a diverse shrub community. Others found mycorrhizal inoculated clonal propagation may be the most effective way to reintroduce ericaceous shrubs (Diaz et al. 2006).

These data demonstrated that different revegetation strategies produced variable results in tree structure (e.g. density, cover). Although sites where container grown pine saplings were planted better matched reference site densities regardless of disturbance type, seeded sites produced densities 2 to 3 times greater than planted sites resulting in a monoculture of pines. Pine densities in sites where container grown pine seedlings were planted were not much different from pine densities reported in other dwarf pine plains studies (see Buchholz and Good 1982).

Pine tree height and basal area increased with time as older sites had taller trees with wider root collars. However, the relationship between age and foliage cover was less obvious. Restoration sites with foliage cover comparable to natural reference sites may be more influenced by fertilizer or site preparation than age. For example, site 1993A, one of two sites tilled and fertilized, was the only site having comparable canopy cover as the reference site. In an earlier study of pine plain restoration, the use of soil amendments

also demonstrated improved recovery of pine tree growth (Fimble and Kuser 1993). Thus foliage growth may be enhanced with the initial use of fertilizer.

After 16 years, the pine-to-oak ratio in restoration sites (12.5:1) was considerably different from reference plots (1:1). Gill (1975) observed a pine-to-oak ratio of 1:2 in dwarf pine plains communities. Although Gill noted that tree abundance in the pine plains is highly variable and a function of fire history, pines in restored sites clearly outnumbered oaks which in some cases failed to establish at all. There are many factors that contribute to the success of establishing oak species and given the highly degraded oligotrophic soils of disturbed pine barren sites, oak survival in these landscapes can be challenging. For disturbed dwarf pine plains landscapes, neither oak seeding nor container grown transplants were an effective technique for establishing oaks on restored sites. Although oaks were not entirely absent from some restored sites there were no differences in oak densities between sites where oaks were either planted or naturally recruited. Factors that influence germination and survival include but are not limited to favorable soil conditions (Korstian 1927; Rao 1988), herbivory (McPherson 1993; Bonfil 1998), timing of planting (Richardson-Calfee et al. 2004) size of transplant (Struve et al. 2000) or even size of container (Burkhart 2006). These reports suggest that additional studies of oak survival in pine barren habitats may be warranted.

Even after 10 to 20 years of restoration at WGR, no revegetation strategy replicated the plant community composition and structure observed in a reference system. Although species diversity of the oldest restored site was similar to the reference site, shrub and oak cover and density differed. However, presence of shrub and herbaceous species in the oldest sites suggest that natural succession may be the most important

factor in restoring the shrub and herb sere if shrub and forb reestablishment are not part of the revegetation strategy. Alternatively, succession may be more influenced by disturbance patch size. For example, sites 1988 and 1993C (where shrubs and oaks were present) were narrow sites no wider than 5 meters giving them a greater forest edge. This greater edge effect may have enhanced recruitment of vegetation from the neighboring forest (see Honnay 1996). Thus, it is logical to assume that smaller disturbed patches, as well as patches closest to an intact forest seed bank, will be colonized sooner and more thoroughly than would larger isolated patches (Bradshaw 1983).

Understanding the disturbance history of a restoration site is crucial for restoration success (Bradshaw 1996). This study included the evaluation of restoration sites that were anthropogenically disturbed (plowing and gravel extraction). Plowing disturbs the upper soul layer without complete disruption of the soil profile while gravel extraction involves the complete removal of upper soil layers leaving only lower soil horizons with very little nutrients and organic matter. Since these mechanical disturbances have different impacts on soil structure, similar restoration strategies may produce different results. The impact of both compaction and top soil removal on restoration outcomes is well documented in habitats with different disturbance regimes (Koerner et al. 1997; Doerr 1998; Holl 2000; Verhagen 2001), especially with regard to mycorrhizal associations (Gould and Liberta 1981; Hutton et al. 1997). In my study it was difficult to differentiate between the effects of time and disturbance type since abandoned gravel pits were the most recently restored and had shorter recovery times. Even so, it was quite possible that age and restoration design may not be as important to recovery success as the ability for surrounding vegetation to colonize restored areas (Holl

et al. 2000; De Souza and Batista 2004). These data suggested, however, that gravel pits tended to be the most dissimilar to other types of restored sites in terms of diversity and tree structure (growth and cover) and that a more aggressive approach to restoration was needed.

This study did not take into account the potential impact of repeated prescribed burning on the pine plains community and was limited to comparing restoration sites to a reference site that last burned by a wildfire in 1959. Comparisons made to reference sites with different fire histories might produce different results (McKessey 2006). However, plant community structure in the 1959 reference site was consistent with that reported in the literature (Lutz 1934; Gill 1975; Good and Good 1975; Good et al. 1998; Buchholz and Zampella 1987) and represented a natural dwarf pine plains community.

The results from this study suggest that several factors are important for the successful restoration of highly disturbed pine plains landscapes. Time, site preparation, and planting method are key considerations to take into account when planning restoration goals and objectives. To better replicate the natural pine plains community, disturbed sites should be planted rather than seeded and site preparation will better enhance growth, survival, and diversity over time. Direct seeding approaches to reforestation have produced variable results in other studies. For example, Torbert et al. (1995) found direct seeding on surface mined Appalachian soil resulted in excessive densities of some tree species and limited establishment of others. Improved establishment of pines and oaks on degraded soils were observed with transplants of nursery grown seedlings (Schoenholtz and Burger 1984; Salifu et al. 2008). In my study, sites planted with pine seedling shad more natural pine tree density when compared with

seeded sites. Similarly, in pine barren wetland habitats, plant cuttings also proved to be a more reliable restoration technique (Mylecraine et al. 2004). Sites planted with rooted cuttings of Atlantic white cedar showed better growth and survival than seeded sites. Although initially more cost effective, these data showed that seeding resulted in an excessive monoculture of pines that may preclude the establishment of other species.

Vegetation structural characteristics (e.g. growth or cover) are often used as surrogates of functional recovery because of their ease of assessment (Allen 1992). Examining the structural complexity of restored sites can indirectly provide information regarding the recovery of ecosystem services and processes (Ruiz-Jaen and Aide 2005a; Ruiz-Jaen and Aide 2005b). Therefore, although I did not directly assess ecosystem function (or measure the return of native fauna), I suggest that reforestation strategies used prior to 1997 fell short of replacing the plant community dynamics observed in surrounding undisturbed dwarf pine plain habitat. Future restoration at Warren Grove Range would benefit from monitoring studies that examine functional recovery of these oligotrophic ecosystems.

In summary, for the short term goal of restoring highly disturbed areas of pine plains habitat at WGR, all restoration strategies produced similar levels of cover but lacked the diversity and structural complexity of the pine plains community. The continued monitoring of restoration projects at WGR and evaluation of new experimental restoration techniques will be of utmost importance for protecting the ecological integrity of the globally imperiled dwarf pine plains community.

IMPLICATIONS FOR PRACTICE

- Restoring mechanically disturbed upland pine barren habitats can be especially challenging due to the extreme oligotrophic conditions of theses soils.
- Planting pine seedlings rather than sowing seeds may better replicate natural pine tree density and growth.
- Time is an important factor for recruitment and recovery of oaks and shrubs into disturbed pine plain habitats. Developing revegetation strategies to accelerate the recovery of these species is warranted.
- Disturbance history should be accounted for in restoration planning as abandoned gravel pits in the pine barrens may require more aggressive revegetation approaches.
- Size and shape of disturbed areas is an important consideration when planning restoration goals as smaller sites with greater exposure to intact forests may recover faster than large sites.
- For pine plains communities, design and implementation of restoration strategies that assist and enhance natural successional processes will be more effective than restorations that only replace absent or missing vegetation.

Table 1: Descriptions of restoration study sites at WGR. (Key refers to location on Figure 1).

Site Key	*Site ID*	*Historic Land Use*	*Area (acres)*	*Year Abandoned*	*Year Restored*	*Revegetation Method*
1	1988	Target Site Lines	1.23	1970's	1988	Planted with pines
2	1993A	Target Area	2.1	1970's	1991 to 1993	Amended and planted with Switchgrass (1991). Tilled and planted with pitch pine and oak seedlings (1993)
3	1993B	Target Area	0.7	1979-1984	1993	Tilled and planted with pitch pine seedlings
4	1993C	Road	0.3	Unknown	1993	Tilled and planted with pitch pine seedlings
5	1996A	Gravel Pit	0.4	1970's	1996	Amended, tilled, planted with pitch pine and oak seedlings. Seeded with oaks.
6	1996B	Gravel Pit	0.25	1951	1996 to 1997	Seeded with pitch pine and oaks (1996) Planted with pitch pine seedlings (1997)
7	1996C	Target Area	0.8	1980's	1996	Seeded with pitch pine and oaks
8	1997	Gravel Pit	0.3	1976	1997	Tilled and planted with pitch pine and oak seedlings

Table 2. Structural characteristics for each of the restoration sites and the reference site. Values are the means ± S.E. Mean values in rows not sharing same letter are significantly different at $p = 0.05$.

	1988	*1993A*	*1993B*	*1993C*	*1996A*	*1996B*	*1996C*	*1997*	*Reference*
pine tree density (per $50m^2$)	13.9 ±1.8 a	33.3 ±8.3 b	31.3 ±5.6 b	33.5 ±5.5 b,c	58.0 ±11.0 c	102.0 ±22 d	102.7 ±2.4 d	31.0 ±8.9 b	37.3 ±4.4 b
% pine tree cover (ground)	65.4 ±8.0 a,b	80.7 ±11.0 a,b	73.5 ±2.0 a,b	76.3 ±5.0 a,b	56.7 ±14.0 a,b,c	45.7 ±1.0 a	89.9 ±3.0 b	26.5 ±5.0 c	62.6 ±5.0 a,b
% pine tree cover (foliage)	60.3 ±9.5 a	105.8 ±15.4 b,c	67.9 ±7.8 a,c,d	78.5 ±1.2 a,c	58.7 ±11.1 a,d	38.2 ±2.2 a,d	48.5 ±8.2 a,d	26.0 ±10.9 d	135.5 ±10.9 b
% pine basal area	0.10 ±0.02 a,b	0.16 ±0.03 b	0.12 ±0.02 a,b	0.1 ±0.01 a,b	0.1 ±0.0 a,c	0.0 ±0.01 a,c	0.1 ±0.0 a,b,c	0.0 ±0.0 c	0.2 ±0.02 b
pine tree height (m)	1.66 ±0.07 a	1.93 ±0.04 a	1.28 ±0.05	1.03 ±0.05	0.30 ±0.02 b	0.24 ±0.01 b	0.59 ±0.02	0.34 ±0.01	1.93 ±0.04 a
oak tree density (per $50m^2$)	1.6 ±1.2 a	0.0 a	0.7 ±0.3 a	8.0 ±0.0 a	3.3 ±2.0 a	4.0 ±4.0 a	0.0 a	4.3 ±0.3 a	39.7 ±12.1
% oak tree cover (ground)	1.4 ±0.8 a	0.0 a	0.0 a	0.0 a	0.0 a	0.0 a	0.0 a	0.0 a	18.0 ±0.5
% oak tree cover (foliage)	0.3 +0.3 a	0.0 a	0.0 a	5.6 +2.1 b	0.2 +0.2 a	0.2 +0.2 a	0.0 a	0.4 +0.1 a	44.7 +14.4
% oak basal area	0.0 ±0.0 a	0.0 a	0.0 ±0.0 a	0.01 ±0.0 b	0.0 ±0.0 a	0.0 ±0.0 a	0.0 a	0.0 ±0.0 a	0.03 ±0.01 b
oak tree height (m)	0.30 ±0.1 a		0.26 ±0.0 a	0.58 ±0.1	0.19 ±0.0 a	0.15 ±0.0 a		0.18 ±0.0 a	1.29 ±0.0
% shrub cover	1.3 ±0.1 a	1.4 ±0.1 a	0.0 a	0.0 a	0.0 a	0.0 a	0.0 a	0.0 a	50.5 ±0.1
% forb cover	1.0 ±0.5 a	1.0 ±0.6 a	6.6 ±2.3	0.0 a	0.0 a	0.0 a	1.1 ±1.0 a	0.0 a	1.7 ±1.1 a
% Total Cover	67.5 ±7.2 a,b	81.8 ±10.1 a	75.0 ±3.0 a,b	76.3 ±4.6 a,b	70.1 ±7.0 a,b	45.7 ±1.0 c,b	89.9 ±1.7 a	26.5 ±5.1 c	86.4 ±3.9 a
Simpsons Diversity	0.62 ±0.04 a	0.37 ±0.11 b	0.23 ±0.1 b,c	0.47 ±0.15 a,b	0.10 ±0.06 c	0.06 ±0.06 c	0.06 ±0.04 c	0.24 ±0.06 b,c	0.60 ±0.04 a

Table 3. Complete plant inventory for all study sites. Presence or absence of plant species was conducted at multiple times throughout the growing season.

					Project				
Species	Reference	1988	1993A	1993B	1993C	1996A	1996B	1996C	1997
Total number of species (richness)	13	16	10	10	10	4	3	4	3
Trees									
Pinus rigida	X	X	X	X	X	X	X	X	X
Quercus coccinea	X								
Quercus ilicifolia	X	X	X	X	X	X	X		X
Quercus marilandica	X	X	X	X	X	X	X		X
Shrubs / Sub-Shrubs									
Arctostaphylos uvaursi		X	X		X			X	
Chimaphila maculata		X							
Gaultheria procumbens	X	X		X					
Gaylussacia baccata	X	X		X					
Gaylussacia frondosa	X								
Hudsonia ericoides	X	X	X	X	X	X			
Ilex glabra		X							
Kalmia latifolia	X								
Lyonia mariana		X							
Pyxidanthera barbulata		X							
Smilax glauca	X								
Vaccinium pallidum	X	X	X	X	X				

Table 3 (continued)

		Project							
Species	Reference	1988.0	1993.1	1993.2	1993.3	1996.1	1996.5	1996.6	1997.0
Forbs									
Carex pensylvanica	X	X	X	X	X				
Dichanthelium acuminatum		X	X	X	X				
Euthamia tenuifolia								X	
Hypericum gentianoides					X				
Melampyrum lineare	X	X	X						
Schizachyrium scoparium		X	X	X	X			X	

Table 4. Percent similarity matrix based on Bray-Curtis distance coefficients of species abundances. (*) denotes significant differences between the two compared sites. Calculated p values are from MRPP pair wise comparisons of abundance data and were adjusted to $p = 0.006$.

	Ref	*1988*	*1993A*	*1993B*	*1993C*	*1996A*	*1996B*	*1996C*
1988	21% *							
1993A	37% *	52%						
1993B	36% *	50% *	75%					
1993C	44% *	46% *	68% *	73% *				
1996A	55% *	33% *	63%	67%	69%			
1996B	60% *	22% *	45%*	46%	49%	71%		
1996C	57% *	22% *	45%	45%	45%	68%	86%	
1997	38% *	45% *	59%	70%	72%	67%	47% *	43%

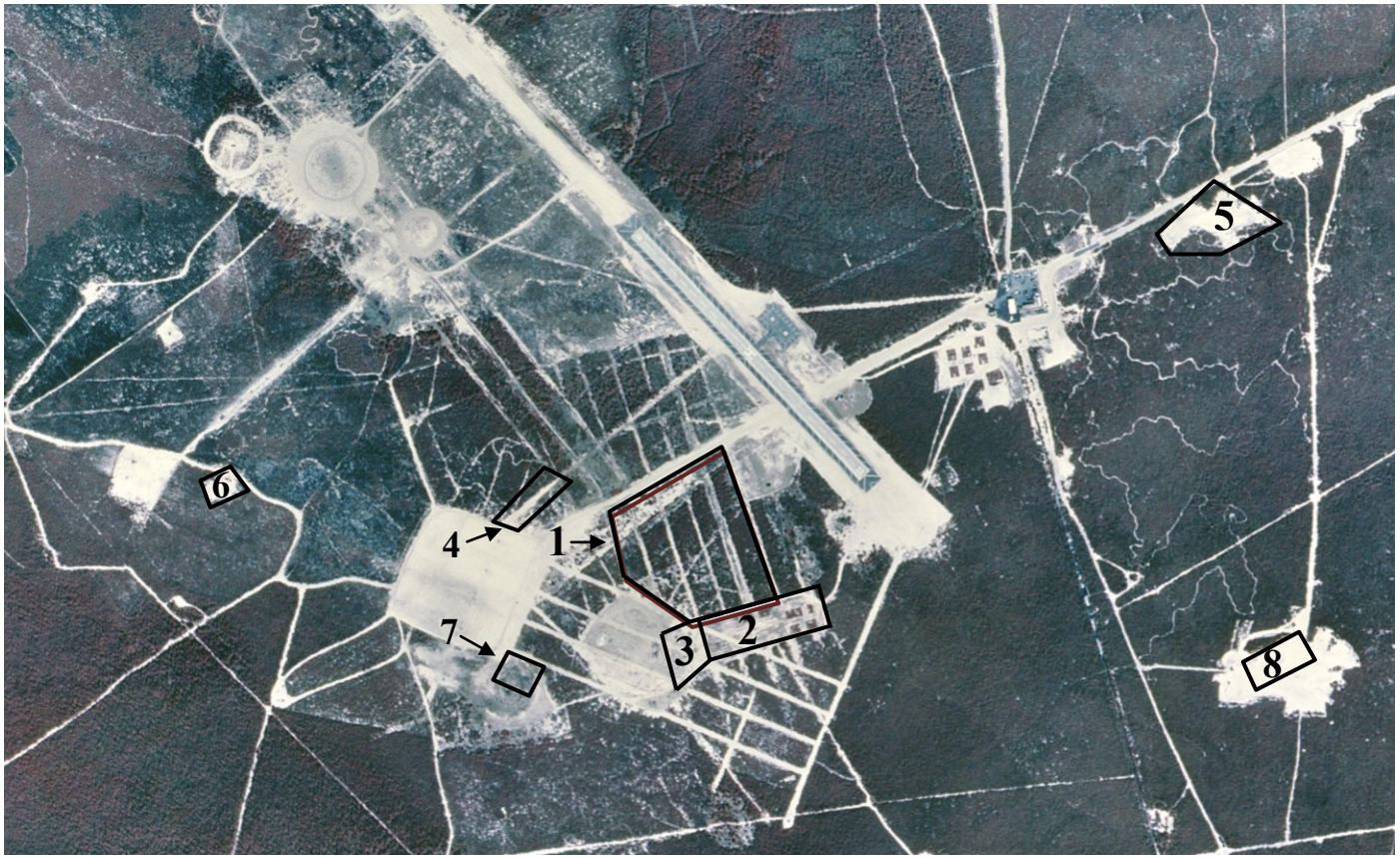


Figure 1. Aerial photo of WGR target area indicating locations of restoration project. The target area occupies approximately 223 ha of the 9,416 ha Air National Guard installation. This area has been impacted by air to ground targeting, site line clearing, road construction, and sand or gravel extraction.

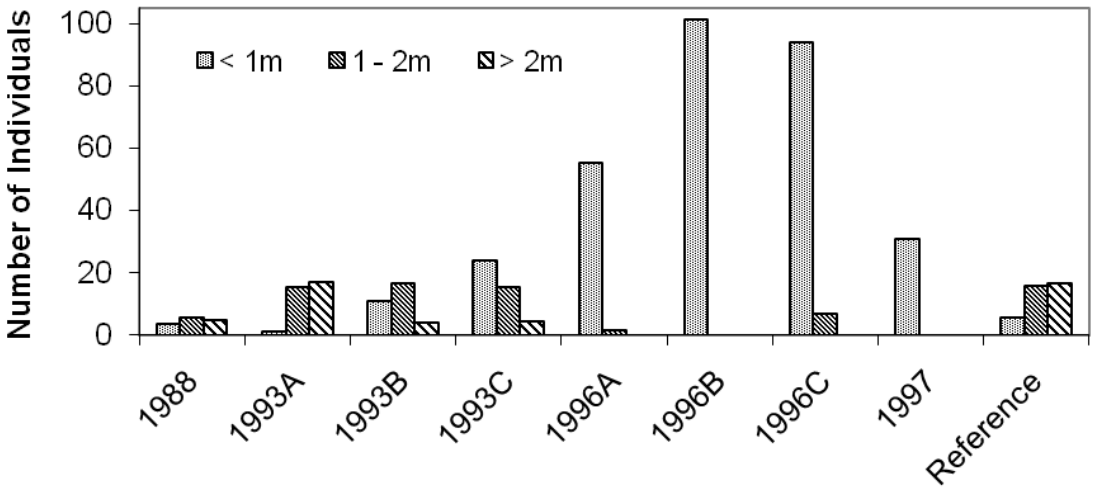


Figure 2. Mean number of pine trees (per 50 m^2) by size class for each restoration site and the reference site.

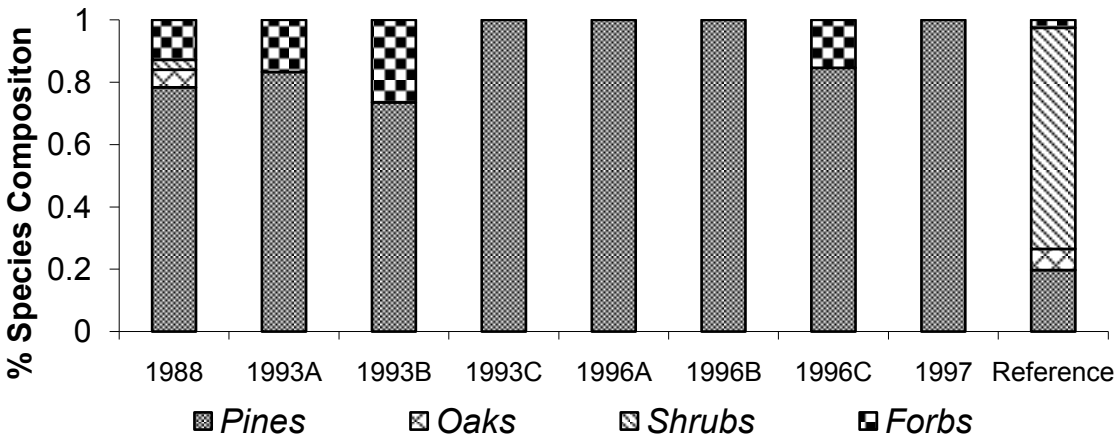


Figure 3. Percent species composition (pooled by plant strata) as determined from relative frequency of individual species density for each restoration site and the reference site.

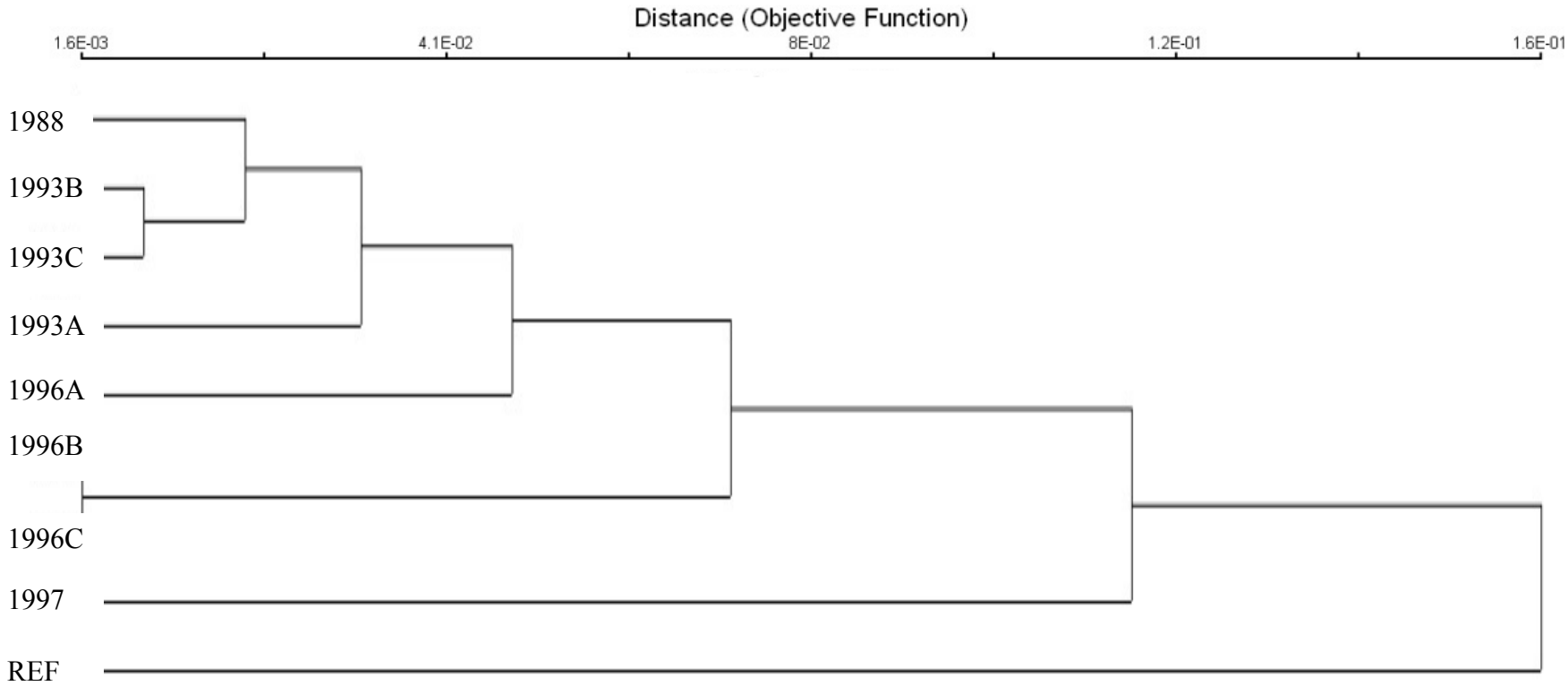


Figure 4. Cluster dendrogram of the restoration sites and reference site based on Bray-Curtis distance coefficients on 14 structural characteristics. See Table 2 for structural parameters and the methods section for a description of the cluster model.

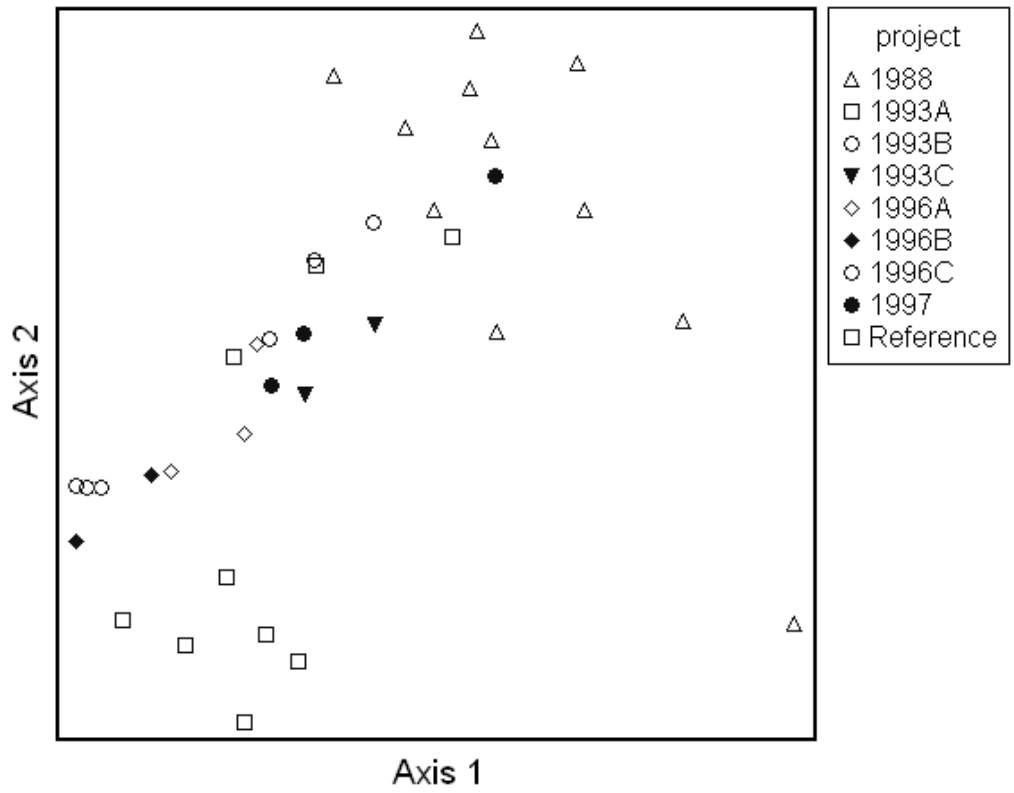


Figure 5. Results of the non-metric dimensional scaling ordination of species abundances per plot for all restoration and reference sites. Each symbol represents a single plot within each study site. Bray-Curtis distance coefficients were used to visualize compositional differences in plant species among sites.

CHAPTER 3: A Case Study of Assisted Succession for the Restoration of an Abandoned Gravel Pit in the New Jersey Pinelands

ABSTRACT

Over 1.7 million ha of land have been impacted by surface mines in the United States. Without restoration, abandoned surface mines can leave the landscape permanently scarred and dysfunctional. Natural recovery of these areas by spontaneous succession is often impeded because mining may leave soils biologically inactive. Restoration is frequently used to reinstate the ecological integrity of these degraded habitats and return them to historic conditions. Guided by natural processes, assisted succession is a restorative intervention that has become a viable alternative to technical or engineered approaches. Assisted succession manipulates the biotic and abiotic factors at a disturbed site enough to facilitate recovery and restore natural succession. In New Jersey, abandoned gravel pits are common and have remained devoid of vegetation for decades. Previous attempts to restore these sites through reforestation resulted in monotypic stands of pitch pine with little natural diversity. In 2001, a highly eroded abandoned gravel pit at the Warren Grove Gunnery Range was restored using soil amendments and native grasses. Within a year after restoration the site showed signs of natural succession comparable to other disturbed areas where spontaneous succession occurred. The purpose of this study was to evaluate the potential use of native grasses to initiate natural recovery processes, thus returning the gravel pit onto a successional trajectory comparable to the surrounding pine plains community. Plant structure and composition of the restored site was compared with that of a 20 year old nearby site recovering through spontaneous succession. Four years following restoration, native grass cover was comparable to the

reference site (25% vs. 29%) and native pitch pine returned to the site at equivalent densities (1000 trees/ha vs. 1080 trees/ha). Both sites were 53% similar in terms of native flora but the gravel pit had a much greater percentage of non-native species. This study demonstrated that assisted succession on abandoned gravel pits in the New Jersey Pinelands may prove to be superior to other methods for restoring natural ecosystem processes in these habitats.

INTRODUCTION

Approximately 1% of the earth's land surface has been impacted by mining (Walker and Willig 1999). Of all the natural resource extraction methods, surface mining has caused the most extensive damage worldwide (Bradshaw and Chadwick 1980). In the United States alone, over 1.7 million ha have been impacted by surface mines (Dobson et al. 1997). Surface mining differs from underground mining in that below ground resources lie at or near the surface and are removed by first stripping the land of vegetation and completely disrupting the soil profile. Since 1977, through the Surface Mining Control and Reclamation Act (SMCRA, Public Law 95–87 Federal Register 3 Aug 1977, 445–532), the United States has mandated that all surface mining projects be restored to a productive and self-sustaining habitat once the resource was exhausted or the mining operations ceased. Over the last few decades, the reclamation of mined lands has become an emerging field within restoration ecology.

Because surface mining causes widespread habitat disturbance, restoring these damaged landscapes can be especially challenging (Bradshaw 1997; Cairns 1999). The factors that limit successful restoration may vary from ecosystem to ecosystem, but are

essentially related to the physical, chemical, and biological damage caused by the mining process (Chambers et al. 1987; Chambers et al. 1994; Wiegleb and Felinks 2001). The overburden, or soil remnants, left behind after mining is physically dysfunctional because it lacks the structural characteristics of undisturbed, intact soil. The remaining soil fraction is typically homogenous and entirely comprised of clay, hard rock, sands or gravel (Bradshaw and Chadwick 1980). These characteristics can lead to increased erosion, increased compaction, or decreased water retention (Conrad et al. 2002). Post-mined soils commonly have high acidic or alkalinic properties, or are contaminated with heavy metals (Wong 2003). The bare mineral soils left behind from mining are also subjected to greater variations in temperature and evaporation rates (Bramble and Ashley 1955; Bell and Ungar 1981). In addition to these physical and chemical constraints, mined soils are nutrient poor and lack organic matter (Davison and Jefferies 1966; Schoenholtz et al. 1992). Nitrogen has often been identified as the critical limiting factor for plant establishment on mine spoils (Bloomfield et al. 1982; Roberts et al. 1988; Schoenholtz et al. 1992). Soil processes (e.g., decomposition) are disrupted because a normal microbiotic soil community is absent (Corbett et al. 1996; Jochimsen 1996). Collectively, these factors impair ecosystem function and inhibit the natural recovery of mined sites. Without any restorative intervention these sites can lack vegetation for long periods of time (Kirmer and Mahn 2001). Restoring surface mined sites requires biologically sound strategies that not only reintroduce native flora but facilitate complete ecosystem recovery without ongoing intervention.

Prior to the enactment of the SMRA, mine restoration was an unregulated practice and revegetation methods were haphazard at best (Holl and Cairns 1994). Mined sites

were either seeded with non-native grasses, converted to plantations, or left to recover naturally (Chambers et al. 1994; Parrotta et al. 1997; Singh et al. 2002). Spontaneous succession, or natural recovery of disturbed habitats without restoration, has been successful in certain mining landscapes where both soil conditions and diaspore availability were conducive to plant establishment (Prach 2003; Rehounkova and Prach 2006). On gravel and sand pits in Europe for example, spontaneous succession occurred when both moisture and local seed were not limited (Rehounková and Prach 2008). However, because of the previously described deleterious effects of mining on soil properties, leaving sites to recover spontaneously has generally been an ineffective means for restoring many landscapes (Bradshaw and Chadwick 1980). An examination of unreclaimed surface mines in Oklahoma showed that even after 70 years of abandonment sites have yet to recover fully (Johnson et al. 1982). Relying on spontaneous succession for restoration can also lead to undesirable outcomes. Abandoned sites are susceptible to invasive species or deviation from historic successional trajectories (Wiegleb and Felinks 2001; D'Antonio and Meyerson 2002). For example, Skousen et al. (1994) identified three separate community types developing on recovering mined land in the Appalachians that differed from historic ones. The successional trajectories for each of these naturally recovering mined sites correlated more with existing edaphic conditions than with the surrounding vegetation. In mined sites where either overburden is biogeochemically dysfunctional, invasive species pose a significant threat, or native seed sources are limited, a restoration strategy other than spontaneous succession is warranted.

When degraded sites have reached a “threshold of irreversibility” (Aronson et al. 1993) and natural recovery is impeded, some restorative intervention is required. Sites

can either be directly restored to a target ecosystem, or a set of restorative steps that assist or direct natural recovery processes can be implemented. The former is referred to as technical or engineered restoration, while the latter is known as manipulated or assisted succession (Prach et al. 2007). The characteristics of these two techniques are not mutually exclusive but represent stages along a gradient of restoration effort. Amelioration of poor soil conditions can often be enough to facilitate natural recovery. This type of assisted succession might involve ripping the soil where site compaction prevents root penetration, fertilizing to improve nutrient deficiencies, replacing sections with top soil obtained from other areas, or adding some native colonizing flora (Bradshaw 1997). At the other end of the gradient, a site may require greater technical measures. This can include improving hydrological conditions by regrading the site to its original contour or transplanting species representative of the target ecosystem. It may also require long term land management. An engineered restoration often aims to bypass primary succession by recreating or directly replacing a "pre-existing state" (Prach and Hobbs 2008). Although technical restorations are costly, a target ecosystem can be quickly attained. As with any highly disturbed site, the desired short or long term outcomes of the restoration will drive the revegetation strategy and the objectives of any project must be clearly developed ahead of time (Bradshaw 1983).

Although much worldwide attention has been given to the restoration of coal or mineral mined lands, less attention has been devoted to landscapes damaged from the extraction of gravel and sand (Price et al. 2005). Also referred to as "borrow pits", these small scale mining operations are created when soil is borrowed from one site and used for fill or construction materials at another site. Despite their small scale, collectively,

borrow pits represent significant land area. In the United States approximately 400,000 ha of land have been impacted by sand and gravel extraction over the last 50 years (Buttleman 1992). At the Savannah River Plant in South Carolina, 1% of the land maintained by the Department of Energy are former borrow pits. A similar scenario is seen in other parts of the world. In the Czech Republic almost 1% of the country's land area has been mined for gravel or sand (Rehounkova and Prach 2008) while in the Darwin Region of Australia, sand and gravel mining destroy approximately 58 ha of native vegetation per year. In many cases these landscapes escape regulatory oversight and are left abandoned.

In New Jersey, sand and gravel mining operations are vital to the state's economy, accounting for almost 50% of the annual domestic value from nonfuel mineral extraction (National Mining Association 1999). Sand and gravel are important aggregates for the construction industry with a major use in cement production and road development. However, sand and gravel operations impact the ecological integrity of the landscape by stripping the land of vegetation, disrupting the entire soil profile and causing habitat fragmentation. Within NJ, an estimated 86% of registered mining operations have been identified as inactive or abandoned (New Jersey Department of Environmental Protection 2007). Although legislative mandates require derelict surface mines to be revegetated with native species, sand and gravel extraction in the US is the least regulated mining industry with 80% of operations falling under state or local jurisdiction (Starnes and Gasper 1996). The ecologically diverse Pine Barrens National Reserve in South Central New Jersey is one area impacted by the effects of gravel mining and at the Air National

Guard Warren Grove Gunnery Range restoration of these highly disturbed areas has become a top priority (New Jersey Air National Guard 2001).

Military operations at the Warren Grove Gunnery Range (WGR) have resulted in a patchwork of disturbed landscapes in the globally rare pine plains habitat (Figure 6). The construction of target and target site lines required the complete removal of vegetation while road and facilities development required fill and gravel to be extracted from local borrow pits. Restoring inactive target sites and abandoned borrow pits become a practical necessity not only to maintain the ecological integrity of the pine plains but to reduce pilot confusion during training exercises. Overall, it has been estimated that 162 hectares of pine plains habitat have been disturbed by military operations. Beginning in 1988, several restoration projects were undertaken at WGR. To restore former target areas, target site lines, and abandoned gravel pits various planting techniques were used. Despite differences in site preparation, all restoration sites were primarily planted or seeded with pines. Evaluation of these sites showed that 20 years after restoration, sites developed into a monoculture of pines and though vegetative cover was comparable to a reference forest many of the plant species in the surrounding intact forest were absent from the restored sites (Chapter 2). Of all mechanical disturbance types, abandoned gravel pits were the least similar to reference systems in terms of plant composition and structure.

Given that the restoration practices applied to disturbed pine plains habitats at WGR replaced vegetative cover but failed to restore the plant composition and structure observed in reference systems, a new approach to restoring these highly degraded landscapes was necessary. These prior technical restorations attempted to replace the

dominant vegetation type (e.g. planting or seeding of pines), but appeared ineffective in restoring underlying abiotic and biotic processes and ultimately ecosystem function, all necessary to support natural recovery.

In spring of 2001 Drexel University's Laboratory of Pinelands Research was contracted by the New Jersey Air National Guard (NJANG) to grade and revegetate a portion of an abandoned gravel pit being used as a helicopter landing zone (HLZ). Warm season native grasses (*Schizachyrium scoparium* and *Panicum virgatum)* were seeded or planted in an effort to reclaim the highly eroded area, restore the environmental integrity with surrounding forest, and provide the military with a useful, safe, open area to perform military exercises. The goal of this project was not to restore the gravel pit back to a dwarf pine plains forest, but to provide a low maintenance training area for the NJANG. The reclamation of the gravel pit (GP) provided an opportunity to research the successional dynamics of upland pine barren habitats. McCormick and Buell (1957) observed that an abandoned agricultural field in the pine barrens was quickly invaded by native herbaceous plants while Little (1998) noted that *S. scoparium* was often the dominate grass in old fields long before the establishment of a woody plant strata. Similarly, I observed the dominance of *S. scoparium* on an abandoned target site (TS) at WGR undergoing natural succession. After observing the similarity between the revegetated GP and the nearby TS, I postulated that, for restoring pine plains habitat, it may be more advantageous to plant native grasses than using traditional reforestation techniques.

The purpose of this study is to evaluate the use of assisted succession for restoring highly degraded pine plains habitats. As an alternative to the reforestation measures

performed in earlier restoration attempts, I hypothesized that planting severely degraded pine plains habitats with native colonizers would be more effective in reestablishing natural processes and subsequently restoring native habitat. Specifically, this study compared the plant recovery patterns of a site restored with native grasses with a disturbed site undergoing natural succession. Because of the extent of gravel mining and other upland disturbances in the New Jersey Pinelands, restoration strategies which facilitate natural recovery are of utmost ecological and economical value.

METHODS

1. Study Area

I conducted this study at the Warren Grove Gunnery Range (WGR) in Burlington County, New Jersey, U.S.A. (lat 39°41'N, long 74°23'W) (Appendix 1). WGR is a military training site occupying 3810 hectares of the 450,000 hectares within the East Plains of the Pinelands National Reserve (PNR). The PNR was designated as the United States first National Reserve and contains one of the largest contiguous tracts of pine barren ecosystems in the world (Forman 1998). The Pine Barrens are characterized by well drained, sandy, nutrient-poor soils, with a high frequency of forest fire that facilitates a unique plant community of fire adapted species (Lutz 1934; Good and Good 1975). A gentle rolling topography influences a mosaic landscape pattern of upland and lowland habitats with high ecological value (Good et al. 1998). Furthermore, it has the largest area of globally imperiled dwarf pine plain forests. The pine plains are a pine-oak forest type where arborescent tree forms are typically absent. Pitch pine, *Pinus rigida*, grows in a diminutive form reaching heights no greater than 3 meters while blackjack oak

and scrub oak (*Quercus marilandica and Q. ilicifolia*) grow in shrub form. Pines have serotinous cones which open only after fire. Vegetative regeneration after fire is typically by basal sprouting (Good et al. 1998). The WGR occupies a patchwork of pine barren habitats that include pine-oak forests, cedar bogs, savannas and a significant area of pine plains.

The pine plains are predominantly underlain by unconsolidated Woodmansie soils which are highly leached and very infertile (Tedrow 1998). Furthermore, these soils are highly acidic, ranging in pH from 3.5 to 5.5. Climate in this region is variable, with both extended summer droughts and periods of heavy rainfall being common (Boyd 2008). Regional annual temperature ranges from 0 to 24°C with annual rainfall of 1100 mm.

The study sites included a reclaimed abandoned gravel pit and a former target site (Figure 6). The abandoned gravel pit was located on the western boundary of WGR while the target site was located approximately 1.0 km to the west. Both sites were located in the active target area of WGR. The gravel pit occupied 2.4 former hectares of pine plains habitat that was cut and then excavated between 1974 and 1976. This site remained devoid of vegetation until 1997 when restoration was performed on a 1.7 acre portion of the gravel pit (GP). This area was planted with nursery grown dwarf pitch pine (*P. rigida*) and black jack oak (*Q. marilandica*) and evaluated in a separate study (Chapter 2). An unrestored portion of the gravel pit was designated by the military as a Helicopter Landing Zone (HLZ). The HLZ occupied approximately 0.81 ha of the gravel pit and by the late 1990's became unsuitable for use due to severe erosion and gullying (Figure 7).

Revegetation of the GP first involved the application of 4 trailer truckloads of aged wood chips spread across approximately 6750m^2 of graded substrate. *Schizachyrium*

scoparium (a C4 perennial warm season tall grass) was then seed broadcasted at a rate of 20 lbs. per acre along with lime and root building fertilizer (10-20-10) into the first 1 - 2cm of soil surface. Mature clumps of *Panicum virgatum* (n=140) were transplanted in two phases (spring and fall) evenly among 7 rows at a 2-meter spacing. An additional 200 *Panicum virgatum* plugs were randomly planted within the rows of mature plants. In fall of 2001, woodchips were redistributed by hand and a second application of fertilizer was applied. The site was then restricted from any military or recreational activity while plants established.

The abandoned target site (TS) included approximately 1.6 ha of pine plains habitat which was bulldozed-scraped between 1956 and 1963. The area was decommissioned in the 1980's. The existing topography along the southern edge of the TS had a swale which created mesic soil conditions. In 1996, a 0.8 acre portion of this inactive target area was chisel plowed and seeded with pine and oaks and evaluated in a separate study (Chapter 2). The remaining area of the TS was left unplanted and natural succession occurred along portions of the swale (Figure 8). Early colonizers included *Hudsonia ericoides*, *S. scoparium*, *Andropogon virginicus*, and *Arctostaphylos uva-ursi*. Plant establishment was attributed to the increased soil moisture retained by the swale and diaspore availability from existing soil seed banks and surrounding forest.

2. Data Collection

To evaluate the effectiveness of restoring disturbed pine plains habitat using native grasses, I measured plant recovery over four years on the reclaimed gravel pit (GP) and compared it with the naturally recovering TS. The TS served as a reference site because it represented an early successional pine plains community recovering naturally

from a mechanical disturbance. I characterized the successional trajectory of each site using above-ground structural characteristics as a proxy for community development. Structural characteristics included: species composition, plant density, cover, growth and productivity.

I conducted plant measurements over four years (2002-2005) on the GP and over three years on the TS (2003-2005). Plant measures were limited to *S. scoparium* during the first two study years (2002 and 2003) but all vascular plant forms were evaluated in the final year (2005). No data was collected in 2004. In 2002, *S. scoparium* density on the GP was obtained by counting all plants within 20 randomly installed 1 m^2 rectangular quadrats. In 2003 density, cover, and growth (plant height) of native grasses were obtained for both the GP and TS from randomly installed 1 m^2 quadrats. Within these quadrats, all plant species were inventoried and *S. scoparium* individuals were counted and measured. I recorded grass height by measuring maximum tiller length of 10 randomly selected plants and measured grass cover by using the line intercept method. Three transects were randomly placed across each quadrat and the total intercept of grasses across the line was divided by the total length of the line (Brower et al. 1998).

In 2005, I assessed total plant community structure (forbs, shrubs, and trees) by randomly installing six 5 x 10 m plots on the GP and TS. Tree and shrub density was measured by counting all individuals within the plots while forb density was measured by counting the number of individuals within two nested 1m^2 subplots. Within these nested sub-plots, I also measured cover, tiller height, and basal diameter of S. *scoparium*. Basal diameter of individual clumps was measured by averaging the lengths of perpendicular axes. I measured height, foliar cover and basal area for all trees within the plot. Tree

measures were limited to *P. rigida* and oak species only. Tree basal area was calculated from stem diameters at ground height (root collar) (Westman & Whittaker 1975). Because dwarf pine trees often have an asymmetrical growth habit, I measured the short and long axis of each tree crown and averaged the two measures to calculate crown diameter and foliar cover (adapted from Brower et al. 1998.). Total ground cover for each plant species was determined with the line intercept method (Bullock 1996) and used to calculate relative cover for each species (Brower et al. 1998). All structural plant measurements were performed in the fall of each study year.

I assessed species composition by inventorying all plant species within the $50m^2$ plot at each site at the beginning and end of the growing season. Although the importance of non-vascular plants has been reported for soil development in pine barren habitat (Sedia and Ehrenfeld 2003), only vascular plants were inventoried in this study. Plant identification followed the nomenclature of Kartesz (1999). I used the New Jersey Higher Plants Manual (Schmid 2001) to determine a plant's native status. I used this plant inventory to calculate species similarity between the two sites for each study year and for each plant strata (forb, shrub, and tree). Non-native plants were identified as a separate stratum. I defined an introduced species as one that had established either intentionally or unintentionally outside its natural geographic range (Snyder and Kaufman 2004).

I determined above-ground productivity (standing crop biomass of *S. scoparium*) by harvesting all clones within 10 randomly selected $1m^2$ subplots after the growing season in the final study of study. Plants were bagged and returned to the lab where shoots were separated from roots, air dried, and weighed.

I collected soil samples from each site to determine soil pH, particle size, and bulk density. Six soil cores were collected at each site with a 5cm diameter soil borer to a depth of 10 cm, stored chilled and taken to the Rutgers Field Station for analysis. Soil pH was analyzed by volt meter on a 3:1 slurry solution of deionized water with dry material. Particle size was determined through sieve analysis and distribution followed the USDA soil classification system (Soil Survey Staff 1999). Soil percolation rates were measured in situ and moisture content was determined gravimetrically by differences in wet and dry weights. Soils were dried for a minimum of 48 hours at 70°C.

3. Data Analysis

I used univariate analysis of variance (ANOVA) to test for differences in soil characteristics, plant density, cover, growth, and biomass between the two sites. To examine temporal changes in *S. scoparium* density and growth, a factorial ANOVA (site x time) was performed for grass data that was collected across multiple years. Assumptions of normality and heterogeneity of variance were tested using the Kolmogoriv-Smirnov Test and the Levene statistic. Square root transformations were performed where data exhibited non-normality. I accepted statistical significance at $p = 0.05$. Similarity of taxa by site and stratum were assessed using the Sorensen coefficient (Brower et al. 1998). Statistics were performed using the SPSS Statistical Software package, Version 15.0 (SPSS 2001).

RESULTS

1. Soil Characteristics

Differences in soil characteristics between the restored gravel pit and the naturally recovering target area are shown in Table 5. The restored site had a significantly higher pH than the reference site ($F_{[1,6]} = 13.4$, $p < 0.05$). Soils from the TS had a higher percentage of silt and fines ($F_{[1,5]} = 30.2$, $p < 0.005$ and $F_{[1,5]} = 23.1$, $p < 0.01$) than the GP.

2. Plant Community Structure

a. Schizachyrium scoparium density and growth

Density of seeded *S. scoparium* varied from year to year. In 2003, *S. scoparium* mean density on the GP was significantly lower than density on the TS (11.9 vs. 23.2 indiv/ m^2; $F_{[1,49]} = 5.34$, $p < 0.05$)(Figure 9). The same pattern was observed in 2005 (16.6 vs. 24.4 indiv/ m^2; $F_{[1,38]} = 5.78$, $p < 0.05$). *Schizachyrium scoparium* density was only measured on the restored site in 2002, but first year levels (21.5 indiv/m^2) were comparable to those observed in TS plots during 2003 and 2005. Despite a slight trend of increasing density from 2003 to 2005 in GP plots, there was no significant differences between years, with both years still below TS levels ($F_{[1,38]} = 5.78$, $p < 0.05$) (Figure 10). Similarly, density did not change between 2003 and 2005 on the TS.

In 2003, the mean maximum tiller height of *S. scoparium* was higher on the TS (18.1 cm) than on the GP (18.1cm vs. 12.5 cm; $F_{[1,46]} = 7.98$, $p < 0.01$)(Figure 11). However, in 2005 there were no differences in height between the two sites ($p = 0.161$). Differences in the amount of ground cover for *S. scoparium* were variable between sites for 2003 (Figure 12). Ground cover for *S. scoparium* on the GP and TS in 2003 was 13%

and 44% respectively ($F_{[1,49]} = 20.6$, $p < 0.001$). In 2005, ground cover for both sites was more similar. Ground cover increased to 25% on the GP and decreased to 29% on the TS ($p = 0.502$). In 2005, there were no differences in mean basal diameter between individuals of *S. scoparium* on the GP and the TS (5.4cm and 6.4 cm; $p = 0.14$) (Figure 13). There was no difference ($p = 0.49$) in standing *S. scoparium* biomass on the GP (0.67 Kg/m2) and the TS (0.29 Kg/m^2) in 2005 (Figure 14).

b. Tree density and growth

Density, height, cover (foliage and basal) of trees (pines and oaks) were measured for both sites in 2005. Pine density did not differ ($p = 0.853$) between the GP (1000 trees/ha) and TS (1080 trees/ha) and while the number of oaks on the GP was lower than the TS, variability on TS was high and the difference in density was not significant (80±49.0 trees/ha and 440±348.7 trees/ha; $p = 0.337$) (Figure 15). Trees on the GP were significantly shorter than trees on the TS. Mean pine tree height on the GP was 0.67m and 1.18m on the TS ($F_{[1,50]} = 11.6$, $p < 0.001$). Mean oak tree height was 0.14 m on the GP and 0.54m on the TS ($F_{[1,11]} = 5.78$, $p < 0.05$) (Figure 16).

Both pine foliage cover and pine basal area were significantly greater on the TS compared to the GP. Mean foliage cover per pine was 0.46m^2 on the GP and 1.1m^2 on the TS ($F_{[1,50]} = 12.1$, $p < 0.001$) resulting in a total foliage cover of 4.9% and 10.5% (Figure 17). Like pine foliage cover, pine basal area on the TS was also greater than the GP (0.002 m^2 vs, 0.0006 m^2 ; $F_{[1,50]} = 7.67$, $p < 0.01$)(Figure 18). Oak basal and foliage area of oaks were negligible for both study groups, each measure contributing less than 0.01% of corresponding cover, and are therefore not reported.

c. Total plant cover

There were no differences in total plant cover between the restored and reference site (Figure 19). Total plant cover was 22.9% on the restored GP and 36.3% in the naturally recovering TS. Figure 20 shows the contribution of cover for each plant stratum. In both the restored and reference sites, grasses accounted for the greatest cover (78.5% and 57.9%) followed by trees (12.1% and 36.8%). There were no differences in cover between the GP and TS for each plant stratum with the exception of forb cover (Table 6). Forb cover was significantly greater on the GP than the TS ($F_{[1,8]} = 8.0$, $p < 0.05$)(Table 6). Although shrub cover on the TS was ten times greater than the GP, the difference was not significant (p= 0.212).

3. Species composition

Between 2002 and 2005, a total of 62 different plant species were identified through comprehensive plant surveys (Tables 7 and 8). In all years, a greater number of non-native species was identified on the restored site. Excluding non-natives, species richness was still higher on the GP than the TS for each plant stratum (Figure 21). There was a net increase in species richness on the GP between 2003 (n = 21) and 2005 (n = 26) while richness remained constant on the reference site (n = 17). Although the number of non-native plants on the GP remained unchanged between 2002 and 2003 (n=11; Table 8), there was a high turnover rate. Similarity in non-native plants between years was 18% (Sorensen). Between 2003 and 2005, the number of non-native species increased from 11 to 17 species with 21% similarity. Total species similarity between the GP and TS was 60% in 2002 and 53% in 2005. Similarity in plant species by stratum differed for each year (Table 9).

DISCUSSION

Previous restoration studies at WGR suggested that restored pine plains habitats had not yet developed the structural complexity observed in intact pine plains forests. Although complete recovery may take many decades, reforested sites at WGR appeared to be developing into mono-dominant stands of pitch pine (*P. rigida*). Because a developing understory is lacking in these restored sites, earlier reforestation attempts fell short of restoring functional processes and facilitating natural recovery. Thus, identifying an alternative restoration approach is vital to the recovery of this disturbed, globally rare landscape.

The purpose of this study was to evaluate a restorative technique that would facilitate natural recovery. The theory was that by using native grasses to catalyze successional processes, disturbed sites could be placed onto trajectories ultimately leading to a more natural state of functional and structural complexity. In this study, I found that despite differences in disturbance regimes, the restored gravel pit and naturally recovering target area shared early successional structural and compositional attributes. However, determining whether these sites fully develop into "mature" pine plain habitats may be difficult to predict and will likely require long term monitoring.

The highest densities of *S. scoparium*, the seeded perennial bunchgrass, were observed on the GP during the first year of establishment, but decreased the following year. Although density was not measured on the TS in 2002, first year densities on the GP were comparable to those observed on the TS in 2003 and 2005. In 2003, however, density significantly decreased on the GP. An increasing trend in density was observed

from 2003 to 2005 and may represent a population recovery even though the rise was not statistically significant. Differences in *S. scoparium* abundance between the two sites may also have been influenced by the differences in site age. Vegetative reproduction of *S. scoparium* occurs when the collection of ramets comprising an individual form hollow crowns which then fragment to produce new individuals (Briske and Derner 1998). The TS may have a greater number of plants because it is older and individual clones may have fragmented into new individuals. Thus, given time, it is probable that the GP will follow a similar pattern with age.

The early germination success and subsequent increased mortality of *S. scoparium* on the GP may be attributed to several possible factors. First, the use of fertilizer may have fulfilled the nutrient requirements for germination but may not have affected growth and survival. The use of nitrogen and phosphorous containing fertilizers has been shown to enhance the germination rates of perennial bunchgrass without affecting long term survivorship (Welch et al. 1962; Richardson and Bowers 1987). Mortality may have resulted from competition as nutrients became depleted (Holechek et al. 1982; Anderson and Birkenholz 1983). This may have been a factor on the GP where high numbers of non-native species were found. In a germination study with *S. scoparium*, Richardson and Bowers (1987) observed a similar increase in mortality after the first year in fertilized plots followed by a slower recovery in later years. The recovery of bunchgrass population, similar to that observed on the GP in 2005 may be the result of a developing soil community. A microbial soil community, which is often lacking in mined soils, develops in response to the increasing deposition of organic matter (Zink and Allen 1998). Consequently, as organic matter is broken down, mineralization results in an

increased availability of nutrients for plants. Changes in nutrient availability may be responsible for both the recovery of bunchgrass populations and the establishment of non-seeded plants (Blumenthal et al. 2003). Secondly, high germination rates may have been due to the addition of mulch which provided the seedlings protection from erosive forces and thermal gradients (Ross et al. 1985; McGinnies 1987; Bussière and Cellier 1994). Furthermore, mulch increases moisture availability and provides a stable carbon source for soil microbes (Schoenholtz et al. 1992; Bradshaw 1997; Zink and Allen 1998; Gonzalez-Sosa et al. 2001). Given that the rainfall of spring 2001 was below average, mulch may have been especially important in retaining moisture for germinating seedlings. Fertilizer and mulch interactions are complex and establishing causal relations for these two amendments on germination and growth was beyond the scope of this study.

Growth and cover of the seeded grass was measured two years after planting. In 2003, *S. scoparium* on the GP had not yet reached the same size as *S. scoparium* on the TS. Individuals on the GP were 30% shorter and provided almost 70% less ground cover than that on the TS. By 2005, there was no difference in tiller height between the two sites and both sites had the same amount of ground cover. Furthermore, the basal diameter of individual clumps in 2005 did not differ between the GP and TS. Although there was no difference in structural attributes between *S. scoparium* populations, I would have expected *S. scoparium* biomass to be lower on the GP because there were fewer plants. However, that was not the case and *S. scoparium* standing biomass on the restored GP was comparable to that of the recovering TS. Therefore, an unmeasured structural attribute must have accounted for the similarity in productivity but difference in clonal

abundance. An increase in tiller density (number of tillers per unit basal area) for plants on the GP would increase biomass without necessarily changing tiller height or basal area. Derner et al. (2004) studied a chronosequence of the structural attributes of *S. scoparium* on restored Texas prairies and found that tiller density was greater for more recently restored sites. Tiller density can be related to above and below ground resource availability (Derner and Briske 1999). Biomass may not have differed between the TS and the GP because individual clones were younger and may have benefited from available fertilizer.

The population dynamics of *S. scoparium*, especially in a restored setting, are complicated and related to the interactions of biotic and abiotic factors. Additionally, not all factors have been mentioned here. For example, the role of vesicular arbuscular mycorrhizal (VAM) colonization on *S. scoparium* will influence plant development (Anderson and Liberta 1992; Meredith and Anderson 1992). Factors affecting germination and growth of *S. scoparium* are beyond the scope of this study but are addressed in later chapters (See Chapter 4). Furthermore, since long term data for the recovering TS does not exist, it is difficult to characterize the successional dynamics. It's unclear how the population of *S. scoparium* individuals may have fluctuated over the last 20 years on the TS. Despite this limitation, *S. scoparium* is clearly a dominant colonizer of the TS and growth and development patterns of *S. scoparium* on the restored GP appear to be mirroring those of the TS.

Unlike *S. scoparium* germination success, there was no survival of other warm season grasses planted during restoration. Local transplants of *P. virgatum* did not survive through the first growing season. Transplanting of grasses is often unsuccessful,

however nursery grown stock has a higher transplantation success rate (Kirt 2001). Although *P. virgatum* has been described as an early colonizer of disturbed sites in the Pine Barrens (McCormick and Buell 1957), it was not found on the TS. Its role in ecosystem recovery is unknown but its use in future restoration may still be warranted.

Natural recruitment of other pine barren species on the GP also followed recruitment patterns observed on the naturally recovering TS. *Pinus rigida* successfully established on the restored GP at densities similar to those observed on the TS. Although abundance was comparable, pines on the GP were significantly smaller. Differences in size are likely a result of differences in recovery age between the two sites. It is difficult to ascertain whether restoration accelerated pine recruitment on the GP since it is unknown when pines first started growing, and at what rate, on the TS. In the pine plains of Long Island, Landis et al. (2005) found growth rates of germinating dwarf pines to be approximately 9 cm per year, similar to those observed in earlier greenhouse studies (Good and Good 1975). Pitch pine on the GP appeared to be growing at a faster rate (17 cm / year), but this could be related to the added soil amendments. Pitch pine growth is highly variable and influenced by site conditions, climate, seed source, and existing vegetation (Ledig and Fryer 1974; Kuser and Knezick 1985). Based on published annual growth rates for *P. rigida*, trees on the TS are approximately 11 to 12 years old and therefore established on the site sometime in the early 1990's. Although the exact date of abandonment could not be ascertained, it appears that successful establishment of pines on the TS required at least three years. That being the case, pines appeared on the GP much faster because they were observed within a year after restoration. Therefore,

restoration may have helped to accelerate the recovery time for *P. rigida* when compared to TS.

The reasons natural recruitment times differed between the sites are not entirely clear but could be related to suitable germination conditions or seed availability. In the pine barrens, *P. rigida* typically regenerates after fire by basal sprouting. Recruitment of any new individuals is accomplished by seed dissemination from serotinous cones. After heating, cones open releasing seeds which only travel short distances by wind (Ledig and Little 1998). Seeds can remain viable in the soil for a year but require adequate moisture to germinate (Little 1998). Germination is inhibited by thick surface litter or dry conditions. Pine seedlings can develop in burned stands as little as a year after fire if there is an available seed source. Landis et al. (2005) found the most limiting factor for new pine recruitment in burned stands of the Long Island pine plains to be seed availability. Similarly, Little (1998) noted that pine recruitment in old fields can be delayed when there is a poor seed crop or inadequate seed source. Given that pine seedlings were observed on the GP one year after restoration and that dormant seed unlikely existed on the GP prior to restoration, an adequate seed source was probably available sometime after restoration. In fact, in May of 2002, a 4800 hectare wildfire burned in the eastern portion of the Pinelands likely providing a rich seed source for the restored GP.

In addition to the availability of seed, pine recruitment may be facilitated by *S. scoparium*. Stone (1911) noted that in abandoned fields, pines will grow rapidly in established stands of *S. scoparium*. Pines may not have established on the TS until an abundant stand of *S. scoparium* colonized the site. *S. scoparium* populations could

facilitate pine tree establishment by modifying soil properties (Conell and Slatyer 1977; Angers and Caron 1998; Derner and Briske 2001; Zak et al. 2003). It's difficult to determine when *S. scoparium* first colonized the TS, but if facilitation was necessary, pine recruitment on the TS may have occurred years after abandonment.

Regardless of timing, pine tree recruitment on both the restored GP and the TS appear to be following similar recovery trajectories. Pine densities are similar but smaller on the GP likely due to the shorter recovery time. It remains unclear if the GP is recovering at a faster rate than the TS however, recovery of the GP would not have been possible without restoration.

Oaks first appeared on the GP two years after restoration. Although abundance was negligible, density between the GP and TS did not differ. In 2005 however, oaks on the GP remained seedling-like and no individuals were taller than 15 cm. Because of the short stature even four years after restoration, the fate of oak establishment on the restored GP is still uncertain. Establishing an oak component in cleared or mined pine plains landscapes has been challenging. In previous restoration attempts at WGR, oaks have failed to regenerate naturally, even in field trials where plots were seeded and fertilized (Fimbel and Kuser 1993). Little (1998) also observed that shrub oaks invaded old field pine stands in small numbers and acorns required a build-up of duff in order to germinate. Problems with oak recruitment are not just limited to the Pine Barrens. The natural regeneration of oaks has been noted as a widespread problem (Sander 1971; Abrams 1992) Decades of fire suppression in many fire dependant forests has lead to an absence of oak species. Without a fire regime, invasive tree species establish and inhibit oak regeneration. Restoring oak forests typically involves prescribed burning which

eradicates undesirable tree species and creates conditions suitable for acorn germination (Hubbard et al. 2004; Fulé et al. 2005). When fire is not an option, oak seeding or planting is often performed (Twedt and Wilson 2002; Madsen and Lof 2005). However, in the pine plains I found limited oak survival on restored sites where container grown oaks were transplanted or acorns were seeded (see Chapter 2). While the germination requirements for aborescent oak trees have been investigated, less is known for the shrub oak species (*Quercus ilicifolia* and *Q. marilandica*) that are common in the pine plains. Populations of these shrub oaks are maintained by basal sprouting after fire, but research on what is needed for acorn germination is lacking (Little and Moore 1949; Little 1989). Factors that influence germination and survival of oaks in general include favorable soil conditions (Korstian 1927; Rao. 1988), herbivory (McPherson 1993; Bonfil 1998), mychorrizal interactions (Daughtridge et al. 1986; Langley et al. 2002), timing of planting (Richardson-Calfee et al. 2004), size of the transplant (Struve et al. 2000), and even size of the transplant container (Burkhart 2006). Collins and Good (1987) found that regeneration of aborescent oaks in the pine barrens was associated with litter depth and light intensity gradients. For restored sites in the pine plains, time may be the most important factor required for a developing shrub oak community. Although the potential for acorn germination exists, oaks may require a more developed litter layer to persist.

The GP and TS showed similar patterns of structural recovery in terms of plant cover. Grasses, primarily *S. scoparium,* contributed the greatest amount of relative cover in each site. While trees were generally larger on the TS, there was no difference in tree cover between the two sites. Furthermore, the GP and TS both had a developing shrub component. Shrub structure (e.g. height) was not evaluated in this study, but the greater

plant cover on the TS may again indicate a difference in age between the two sites. Only forb cover was found to differ between the GP and TS. Although forbs were present in the TS, they did not contribute a significant amount to total cover. An increased abundance of the early colonizing forb, *Diodia teres*, was responsible for 2.2% of ground cover seen in the GP. *Diodia teres* was originally considered an introduced weed (Harshberger 1916) but has since been considered naturalized and an early colonizer of disturbed sites in the pine barrens (Boyd 1991). *Diodia teres* was also an early invader of other WGR experimental restoration studies (Fimble and Kuser 1993).

With graminoids accounting for over 50% of the total cover, grasses remained the dominant plant form on both the recovering TS and restored GP. However, the emergence of shrub and tree cover suggests that successional changes are occurring and as these plant forms continue to develop herbaceous cover is expected to decline. The amount of herbaceous cover can be inversely related to the tree foliage cover owing to the effects of light availability to the soil surface (Inouye et al. 1987; Pietrzykowski 2008). Forbs may represent a larger component of total cover on the GP because tree and shrub cover is not as developed as the TS. Alternatively, forb species may be less tolerant of other changes occurring above or below ground (Tilman 1987; Walker and Moral 2009). *S. scoparium* may remain persistent during the shrub and tree development because of its density and tolerance of infertile and xeric soil conditions (Bard 1952; USDA 2002). Given that vegetation was sparse (<50% cover) and still developing on both the TS and GP, these sites likely represent an early successional sere of pine plains forest and appear to be following a similar patterns of recovery.

Similarities in species' recruitment are less pronounced between the two sites than their structural floral attributes. Not only was there a higher number of non-native species found, but there was more native species turnover on the GP when compared to the TS. An increase in the number of native forbs on the GP between 2003 and 2005 contributed to the decreased similarity (Sorensen) between sites. In one of the only studies of succession on a plowed field in the New Jersey Pine Barrens, McCormick and Buell (1957) observed a 56% and 100% increase in grass and forb species over an eight month period of recovery. Making generalizations regarding species recruitment patterns during succession is difficult owing to the fact that succession is dependent on many factors including disturbance type and history, climate, moisture, soil type, nutrient availability, soil community, allelopathy, herbivory, facilitation, and competition (Elton 1958; Connell and Slatyer 1977; Denslow 1980; Tilman 1987). Following disturbance, species' richness can increase over time (Inouye et al. 1987) or decrease over time (Meiners et al. 2002; Wilson and Tilman 2002) but causes for species' richness patterns have long been debated (Mittelbach et al. 2001). Although plant inventories for pine plains communities are not a substitute for chronological studies that document species recruitment, they're useful for identifying species-habitat associations. A review of historic and recent Pinelands floral inventories (Lutz 1934; Forman 1998; New Jersey Air National Guard 2005) indicate both sites have a high number of species associated with disturbed conditions. This may suggest that reaching a successional target may be years away, especially for the GP which had a higher number of non-natives. Determining if species recruitment and extinction patterns on the TS and GP will converge over time is difficult, but the similarity of dominant native species observed at this stage is encouraging.

The most striking difference between the GP and TS was the number of non-native species that germinated on the GP. Thirty-one non-native species (50% of the total species richness) occurred, 11 of which were exotic. Furthermore, because of annual differences in non-native diversity, these species represent a range of life history strategies. Non-natives were not unusual at WGR, but the increased numbers of species found on the GP were high relative to other disturbed areas at the installation (Bien et al. 2009). It was unlikely that non-native seeds were transported from airborne, animal, or mechanical vectors, but rather were mixed with the wood chips imported from a recycling center. The mulch was imported from local producers within the county and likely contaminated with non-indigenous seed from rural yard waste. Mulch produced from pine-oak forests on WGR has since been recommended for future restoration projects to prevent non-native introductions (New Jersey Air National Guard 2005).

Because invasive species are poorly adapted to acidic infertile soils, mulch may have harbored the seed, but other factors supported their establishment. An increase in both soil pH and nutrient supply can make restored sites more susceptible to invasions (Hobbs and Huenneke 1992; Thompson et al. 2001; Gilbert and Lechowicz 2005). The GP was restored using root building fertilizers which provided an adequate supply of nitrogen and phosphorous to a system where nutrients were generally limited. The elevated soil pH on the restored GP further increased its susceptibility to non-native establishment. Pine barren soils typically range between a pH of 3.6 and 5.5 (Tedrow 1998) but soil pH on the GP after restoration was 6.17. It is not clear why soil pH on the restored GP became elevated. Non-native species can alter soil pH, (Ehrenfeld et al. 2001), but the elevated pH on the GP was likely from introduced amendments and not a

result of non-native soil interactions. The decreased soil acidity measured on the GP may have been due to the use of imported rock incorporated into the site during grading. Non-indigenous rock is often used at WGR for road development or other infrastructure projects. This material may contain alkaline feldspar that can buffer acidic soils when weathered (van Breemen et al. 1983; Mol et al. 2003). Invasion susceptibility of future restoration projects at WGR would be reduced if any soil amendments (e.g. mulch and fill) originated from local sources only.

Whether or not non-natives persist on the GP is yet to be determined. If they do remain they can potentially have negative impacts on restoration. Non-native plants can alter natural ecosystem processes and impede natural succession (Kourtev et al. 2002; Levine et al. 2003; Wolfe and Klironomos 2005). Batten et al. (2006) found that invasives changed the soil microbial community in serpentine grasslands enough to give them a competitive edge over native grasses. Invasive species can modify ecosystem process by altering nutrient cycling (Ehrenfeld 2003). Shifts in soil microbial composition and subsequent impacts on nutrient dynamics can impede natural recovery by altering successional processes. It remains to be seen whether non-native species will impact successional trajectories enough to prevent the development of a mature pine plains community. This study indicates that abundance of non-natives is low but further monitoring of the GP is warranted to determine long term outcomes. Should the number of non-natives increase and interfere with recovery and growth of native species, further restoration may be necessary. In this case, prescribed burning may be beneficial for eradicating non-natives as well stimulating native species.

CONCLUSION

Sand and gravel mining in New Jersey is vital to the state's economy, but when abandoned, the mines can fragment the landscape. Although sand and gravel mines found elsewhere in the world have recovered through spontaneous succession, surface mines in the New Jersey Pinelands have often failed to completely regenerate. Without some intervention, abandoned mines in this ecologically rich area can remain absent of vegetation for decades. Prior attempts to restore these degraded areas using reforestation techniques have produced a mono-culture of slow growing native pines with little plant diversity, even up to 15 years after restoration. The purpose of this study was to determine the efficacy of using assisted succession as a method for restoring mined or cleared landscapes in upland forests of the New Jersey Pinelands. Assisted succession is a restorative practice that relies on natural processes more so than engineered or technical restorations designed to accelerate replacement of a historical system. Unlike technical measures which often skip successional seres or bypass succession altogether, the goal of assisted succession is to jump start successional processes enough to restore natural recovery. By ameliorating the environmental conditions preventing natural succession, assisted succession may be a more reliable technique for restoring ecosystem structure and function endemic to the landscape. It may take longer to reach restoration goals using assisted succession, but this strategy is more economical and in the long term more efficient for restoring ecosystem functions.

To ameliorate the highly degraded conditions on an abandoned gravel pit in the pine plains, restoration was performed using mulch and native warm season grasses as an alternative to reforestation. In this study, I found that after four years of restoration, the

gravel pit showed evidence of recovery similar to a site undergoing spontaneous succession. Although it may be several decades before the restored site and the naturally recovering site develop into a mature pine plains community, this study suggests that the community structure and composition of both sites are following comparable successional trajectories.

Table 5. Soil characteristics for each study area (values represent mean +/- 1 S.E.M.), Significance values based on ANOVA.

	Gravel Pit	Target Site	
pH	6.17 +/- 0.4	4.98 +/- 0.0	*p < 0.05*
Bulk Density (g/cm^3)	0.71 +/- 0.2	0.67 +/- 0.2	*NS*
Perc Rate (L/min)	0.33 +/- 0.6	0.25 +/- 0.4	*NS*
Grain Size			
Gravel	15.1% +/- 2.5	12.3% +/- 2.6	*NS*
Sand	84.5% +/- 2.5	85.5% +/- 2.4	*NS*
Silt	0.3% +/- 0.2	1.6% +/- 0.3	*p < 0.005*
Fines	0.1% +/- 0.0	0.4% +/- 0.1	*p < 0.01*

Table 6. Absolute cover values +/- 1 S.E.M. (%) with p values from ANOVA for each plant stratum.

	Gravel Pit	**Target Site**	
Trees	2.8 +/- 1.7	14.0 +/- 7.2	*NS*
Shrubs	0.2 +/- 0.2	2.0 +/- 1.3	*NS*
Forbs	2.2 +/- 0.6	0	$p < 0.05$
Graminoids	18.2 +/- 6.4	22.1 +/- 3.2	*NS*

Table 7. Native plant species inventory for each site for each study year. Presence or absence of plant species was conducted at multiple times throughout the growing season.

	2002	2003		2005	
Species	Restored	Restored	Reference	Restored	Reference
Natives					
Trees					
Acer rubrum		X			
Betula populifolia	X	X		X	
Pinus echinata				X	
Pinus rigida	X	X	X	X	X
Pinus virginiana				X	
Quercus ilicifolia		X	X	X	X
Quercus marilandica		X	X	X	X
Shrubs/Sub-shrubs					
Arctostaphylos uvaursi		X	X	X	X
Comptonia peregrina		X	X	X	X
Gaylussacia baccata		X	X	X	X
Hudsonia ericoides		X	X	X	X
Ilex opaca				X	
Lyonia mariana		X			
Morella pensylvanica	X	X	X	X	X
Vaccinium pallidum		X	X	X	X

Table 7 (continued)

	2002	2003		2005	
Species	Restored	Restored	Reference	Restored	Reference
Natives					
Graminoids					
Andropogon virginicus			X		X
Aristida dichotoma	X	X	X	X	X
Carex pensylvanica		X		X	
Carex tonsa		X			
Cyperus dentatus		X			
Dichanthelium sp.	X		X	X	X
Panicum virgatum	X	X	X	X	
Schizachyrium scoparium	X	X	X	X	X
Forbs					
Diodia teres	X	X	X	X	X
Euthamia caroliniana				X	X
Hypericum gentianoides	X	X	X	X	X
Krigia virginica				X	
Polygonella articulata	X	X	X	X	X
Pseudognaphalium obtusifolium				X	
Trichostema dichotomum				X	
Viola primulifolia				X	

Table 8: Non-native plant species inventory for each site for each study year. (See text for definition of non-native. Species marked with an asterisk are considered exotic species).

	2002	2003		2005	
Species	Restored	Restored	Reference	Restored	Reference
Introduced					
Trees					
Juniperus virginiana		X		X	
Populus grandidentata	X			X	
Shrubs/Sub-shrubs					
Baccharis halimifolia				X	
Parthenocissus quinquefolia	X				
Rhus copallinum		X		X	
Graminoids					
* *Digitaria sanguinalis*	X	X			
Eragrostis spectabilis				X	
* *Setaria viridis*				X	
Forbs					
* *Abutilon theophrasti*				X	
Ambrosia artemisiifolia				X	
Antennaria neglecta var. neodioca	X			X	
Apocynum androsaemifolium			X	X	
* *Centaurea biebersteinii*		X			

Table 8 (continued)

Species	2002 Restored	2003 Restored	2003 Reference	2005 Restored	2005 Reference
Introduced					
Forbs					
Chamaesyce maculata	X	X			X
* *Cirsium vulgare*		X			
Conyza canadensis	X			X	
Epilobium sp.				X	
Erechtites hieracifolia				X	
* *Erodium cicutarium*		X			
Eupatorium hyssopifolium		X			
Hieracium sp.	X				
* *Lepidium campestre*				X	
Lespedeza virginica		X			
* *Medicago lupulina*	X				
Oxalis dillenii	X				
Nuttallanthus canadensis		X			
* *Petunia sp.*				X	
Polygonum pensylvanicum				X	
* *Rumex acetosella*	X				
Solanum ptychanthum		X		X	
* *Trifolium arvense*	X				

Table 9. Percent similarity between the Gravel Pit and Target Site for each plant form and for all plants combined. Values are calculated from Sorensen's coefficient of similarity.

	2003	**2005**
Trees	75%	67%
Shrubs	92%	92%
Forbs	100%	67%
Graminoids	55%	67%
Non-natives	0%	0%
Total Site Similarity	60%	52%

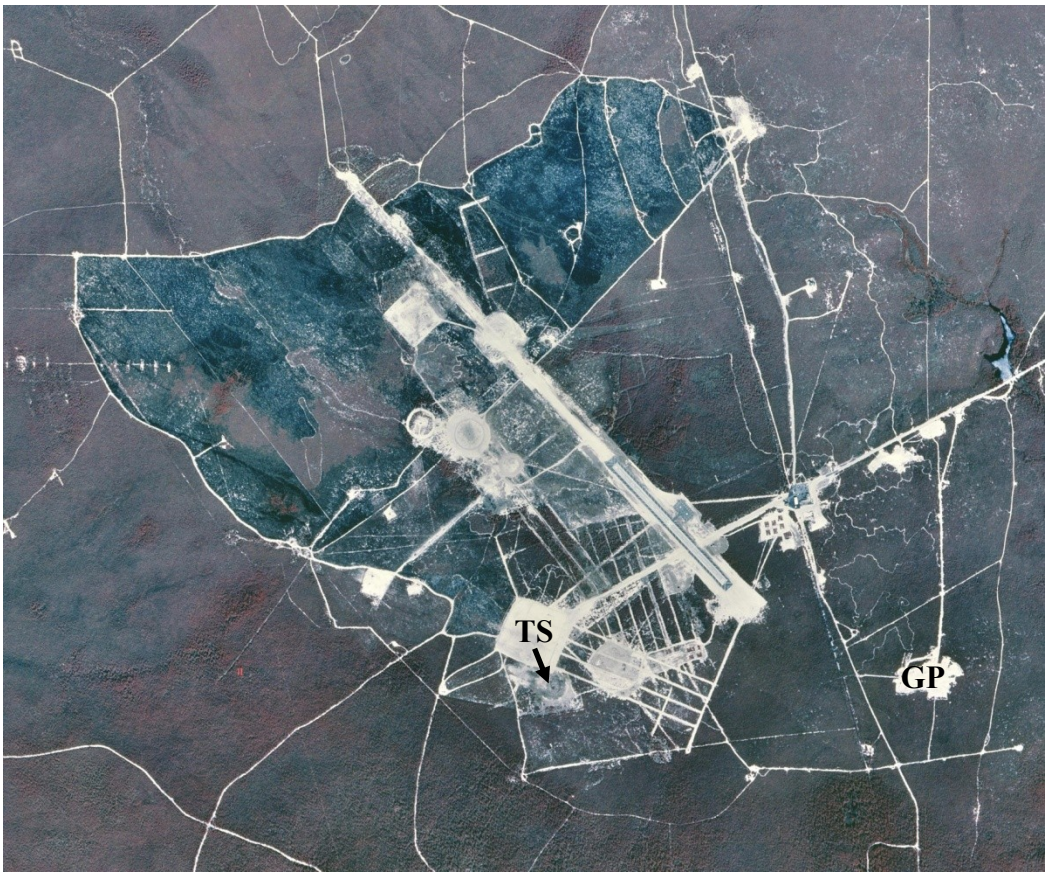


Figure 6. Location of study sites within the active target impact zone of the Warren Grove Gunnery Range. GP = restored gravel pit; TS = target site.

Figure 7. The severely eroded gravel pit. Erosion and gullying of the gravel pit prevented the military from using it as a helicopter landing zone.

Figure 8. The abandoned target site (TS) used as a reference for mechanically disturbed pine plains habitat. This site was abandoned in the late 1980's and recovered naturally by spontaneous succession.

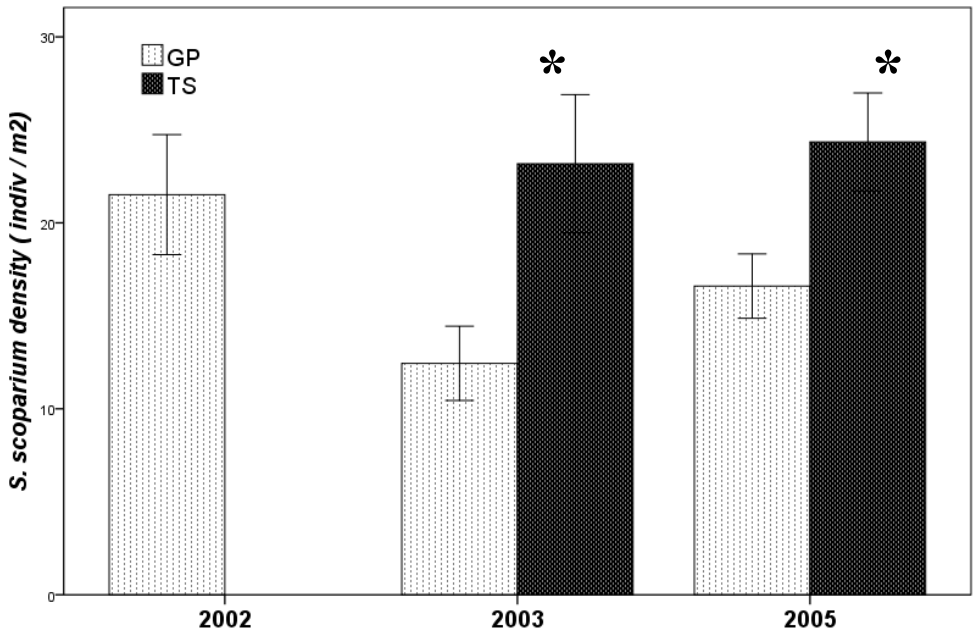


Figure 9. Mean density (individuals / m^2) for *S. scoparium* on the restored gravel pit and naturally recovering site target site for each study year. Error bars represent +/- 1 S.E.M. Significance at $p < 0.05$ is indicated with * above bar.

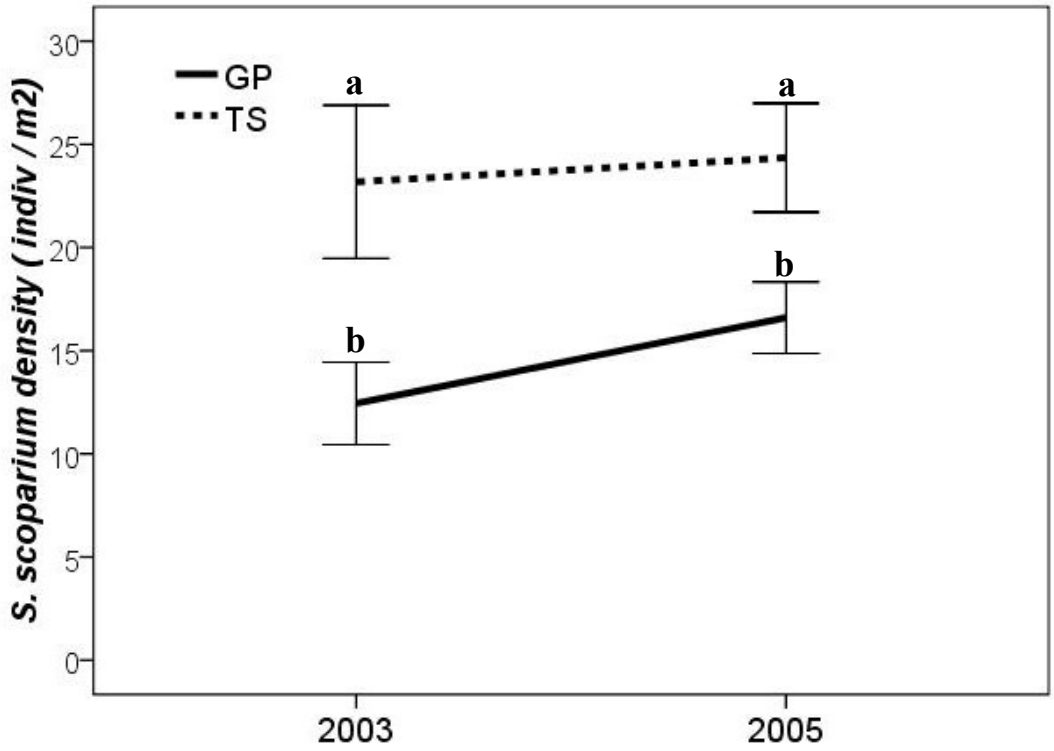


Figure 10. Differences in mean plant density (individuals / m^2) between the seeded *S. scoparium* on the restored gravel pit and the naturally recruited *S. scoparium* on the target site for both study years. Values are the means +/- 1 S.E.M. Data points not sharing same letter are significantly different at $p < 0.05$.

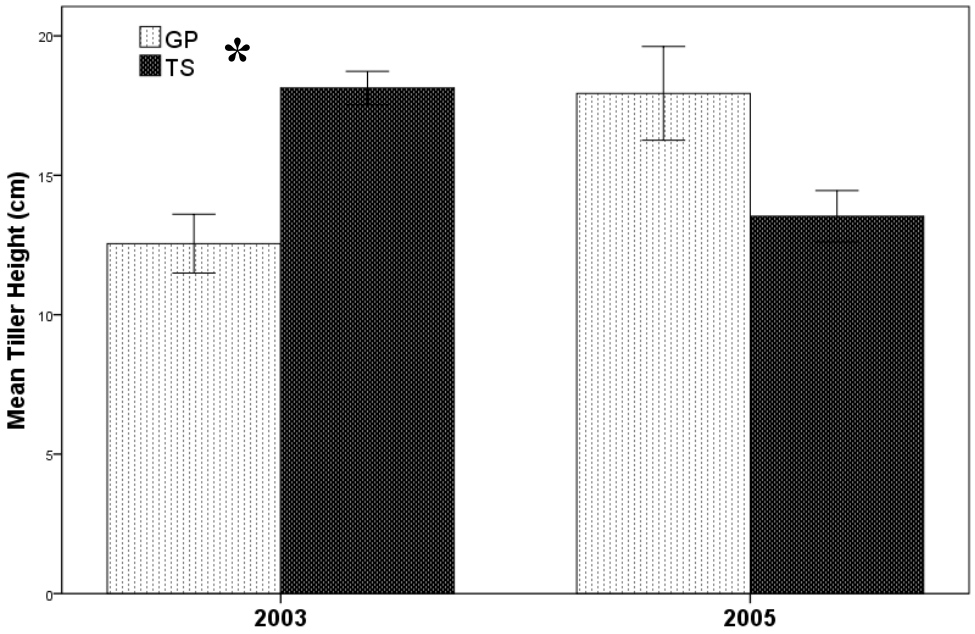


Figure 11. Differences in mean tiller height between seeded *S. scoparium* on the restored gravel pit and naturally recruited *S. scoparium* on the target site for both study years. Error bars represent +/- 1 S.E.M. Significance at $p < 0.05$ is indicated with * above bar.

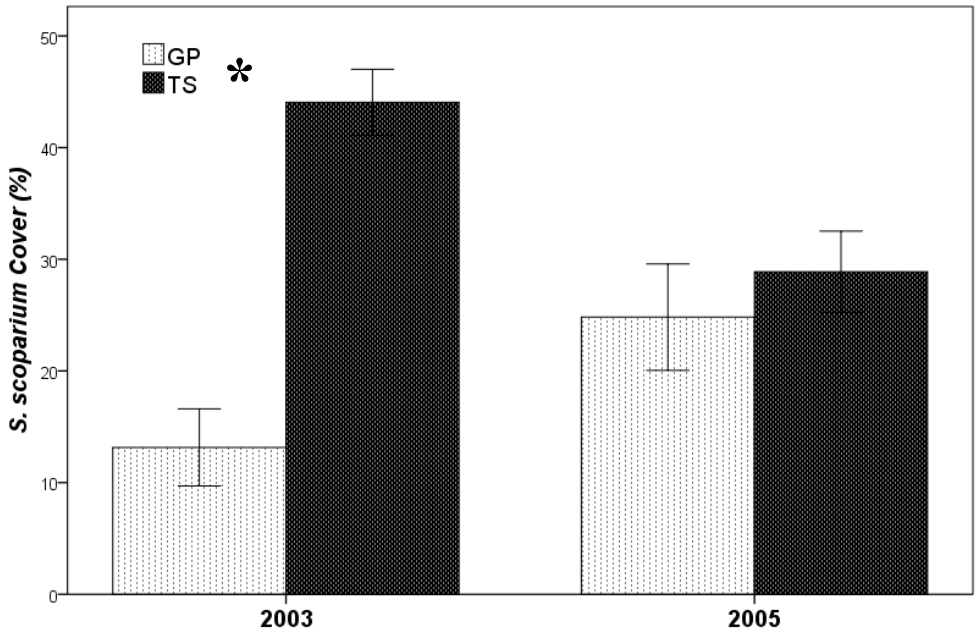


Figure 12. Differences in mean cover between the seeded *S. scoparium* on the restored gravel pit and naturally recruited *S. scoparium* on the target site for both study years. Error bars represent +/- 1 S.E.M. Significance at $p < 0.05$ is indicated with * above bar.

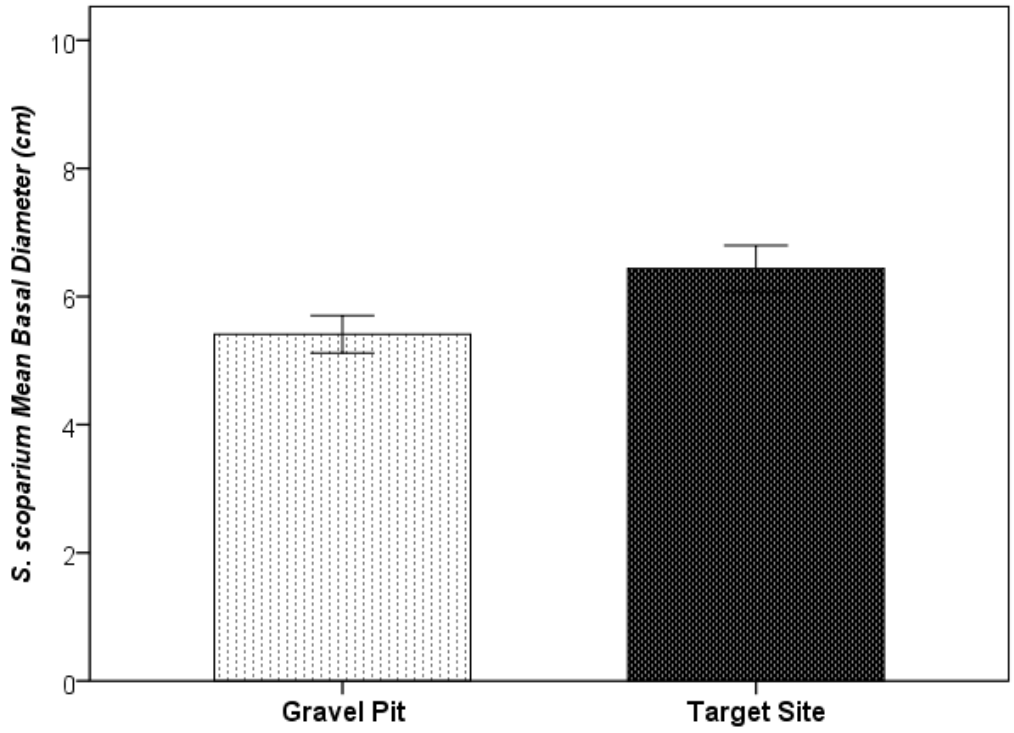


Figure 13. Differences in mean basal diameter between the seeded *S. scoparium* on the restored gravel pit and naturally recruited *S. scoparium* on the target site in 2005. Error bars represent +/- 1 S.E.M.

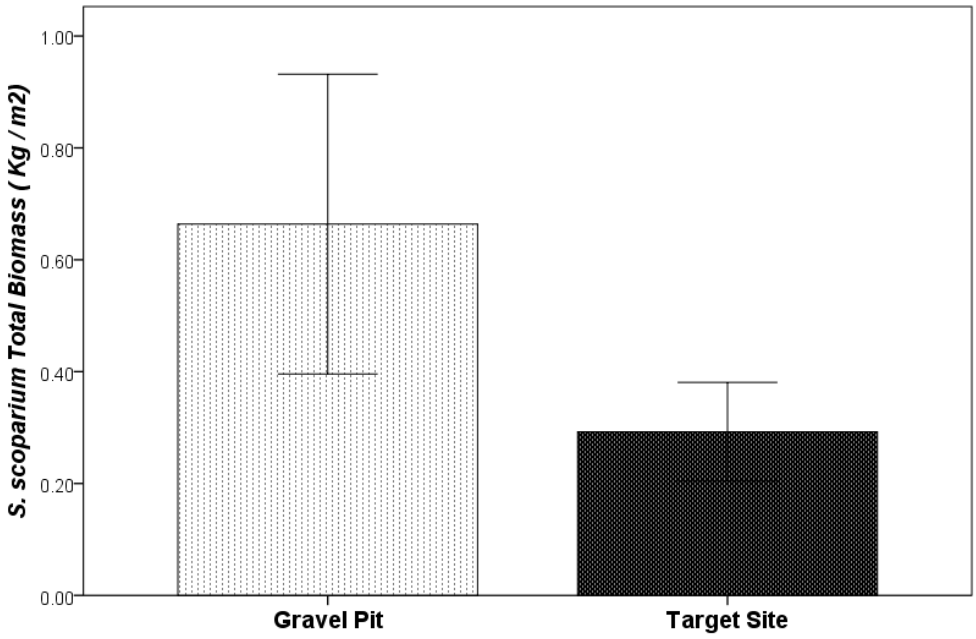


Figure 14. Differences in mean standing crop biomass between the seeded *S. scoparium* on the restored gravel pit and naturally recruited *S. scoparium* on the reference site in 2005. Error bars represent +/- 1 S.E.M.

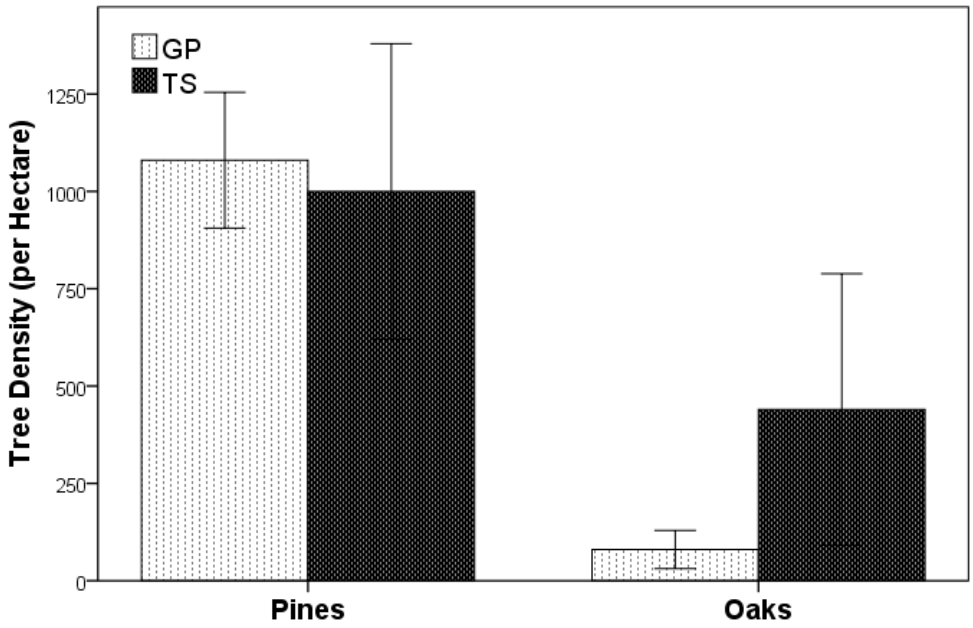


Figure 15. Mean density for pines and oaks on the restored gravel pit (GP) and naturally recovering target site (TS) in 2005. Error bars represent +/- 1 S.E.M.

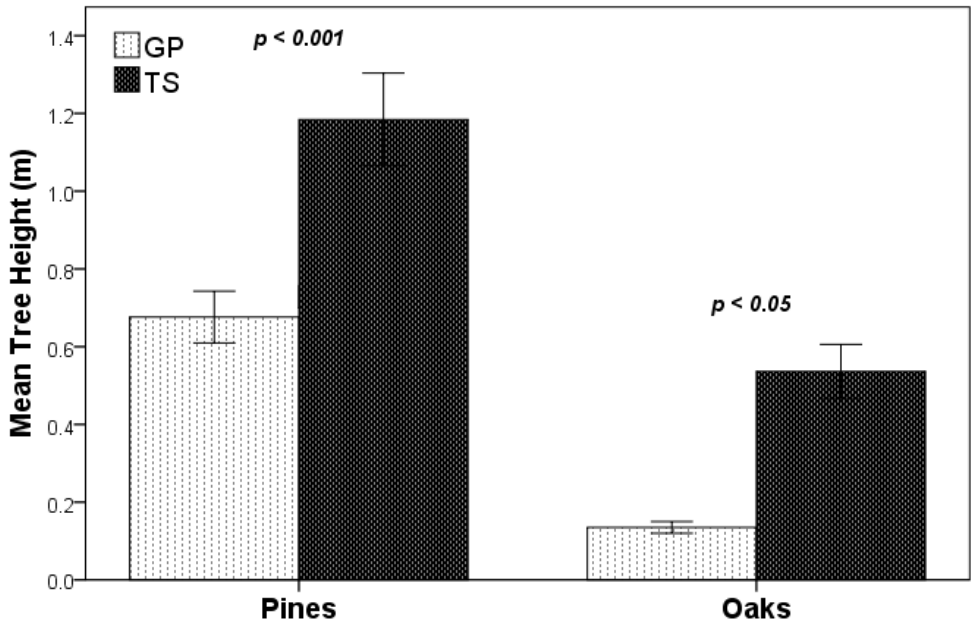


Figure 16. Mean tree heights for pines and oaks on the restored gravel pit (GP) and naturally recovering target site (TS) in 2005. Error bars represent +/- 1 S.E.M. Significance values based on ANOVA.

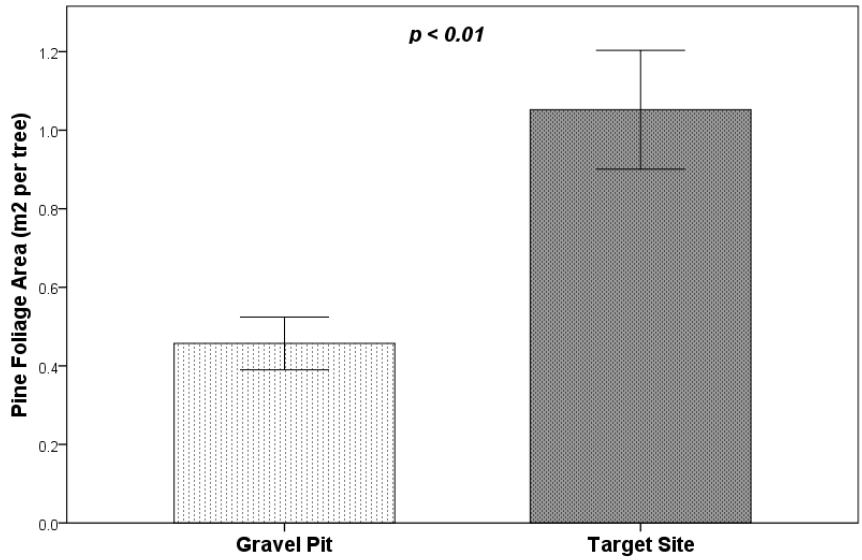


Figure 17. Pine foliage area as measured using crown diameters for each study site in 2005. Error bars represent +/- 1 S.E.M. Significance values based on ANOVA.

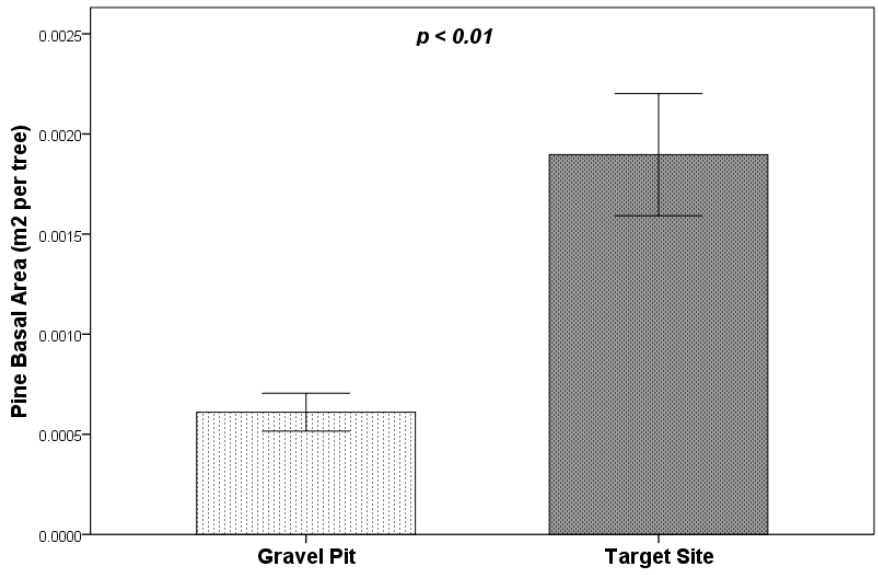


Figure 18. Pine basal area per tree measured using root collar diameters for each study site in 2005. Error bars represent +/- 1 S.E.M. Significance values based on ANOVA.

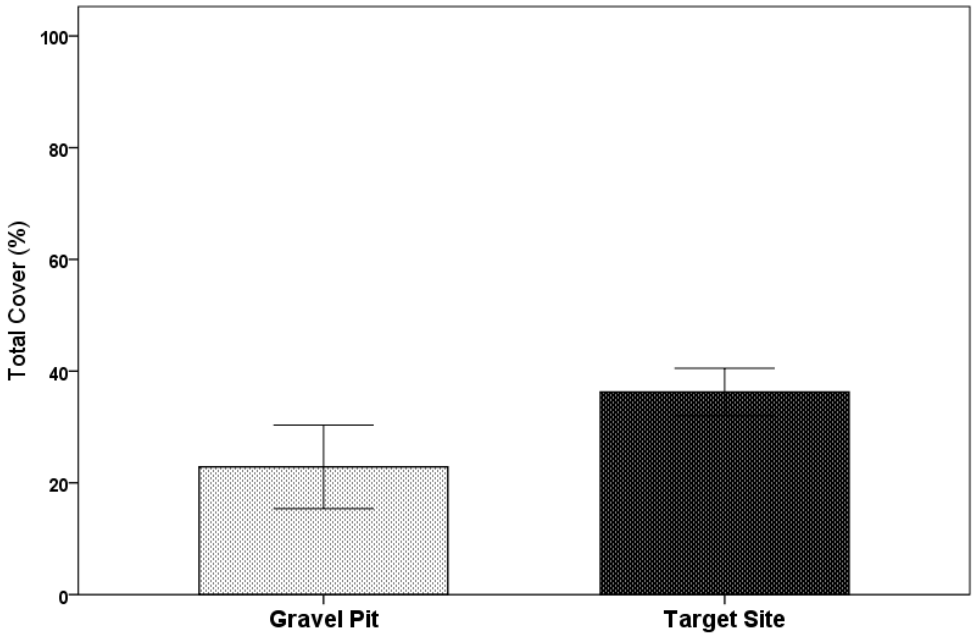


Figure 19. Total plant cover measured on the Gravel Pit and Target Site in 2005 using the line intercept method. Error bars represent +/- 1 S.E.M.

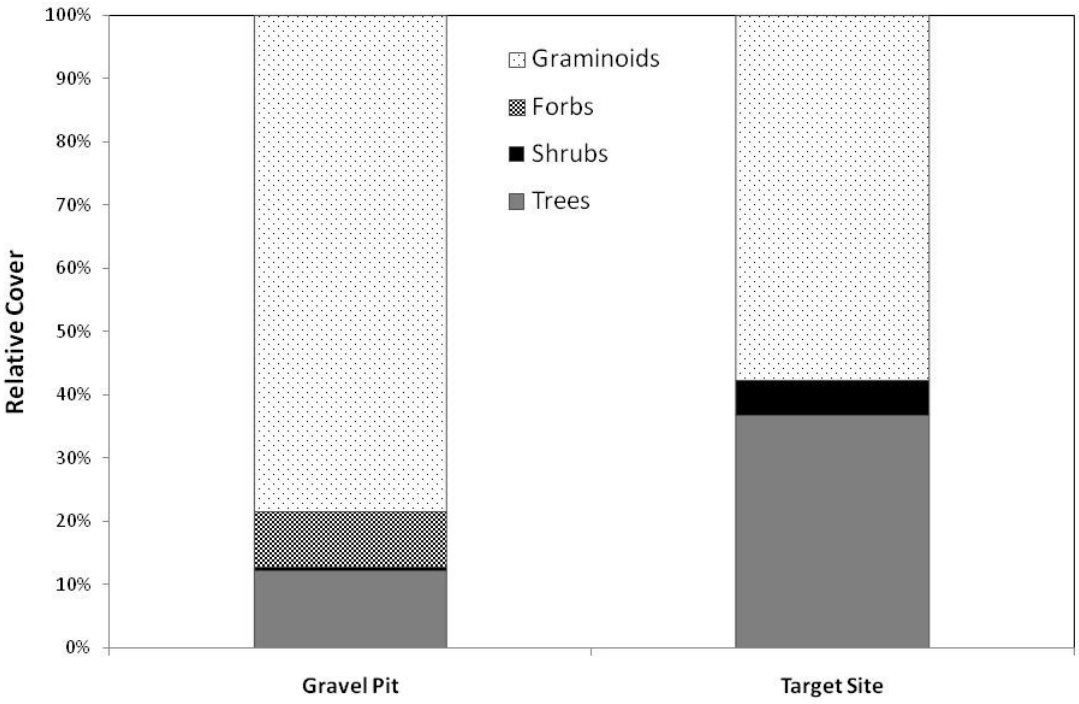


Figure 20. Relative cover (%) for each plant stratum for each site in 2005. Specific differences in actual cover for each stratum are shown in Table 6.

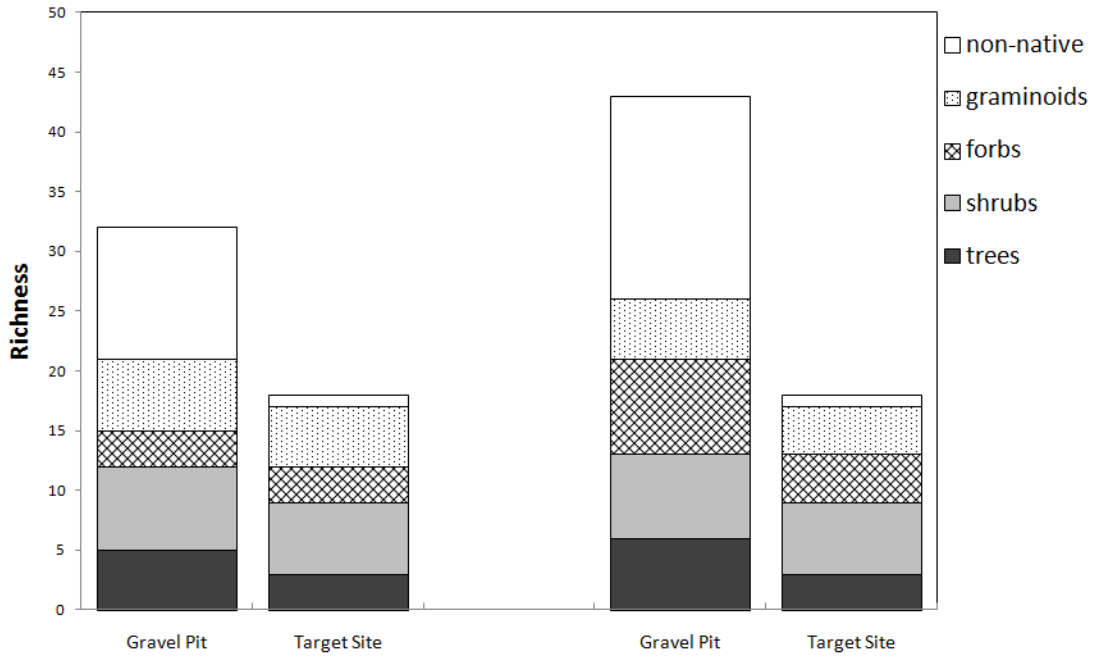


Figure 21. Species richness separated by plant form (stratum) for the Gravel Pit and Target Site in 2003 (left) and 2005 (right).

CHAPTER 4: Factors contributing to successful germination and establishment of *Schizachyrium scoparium* on an abandoned gravel pit in the New Jersey Pinelands

ABSTRACT

The establishment of native plants is paramount for successful restoration. However, because highly disturbed soils have structural problems, dysfunctional nutrient cycles, and an absence of below ground biota responsible for mineralization and nutrient cycling, restoring native plant communities can be difficult. Restoration requires selecting suitable species that can ameliorate the effects of disturbance and re-establish successional pathways. In doing so, ecosystems can recover on their own without further intervention. In the New Jersey Pinelands, abandoned gravel pits often remain absent of vegetation for long periods of time. Identifying revegetation methods that promote recovery of ecosystem function and structure are of great ecological importance for this region. *Schizachyrium scoparium*, a native, warm season, perennial grass, has been shown to be a good pioneer species for severely disturbed upland landscapes. I present here the results of a germination field trial that identified factors that contribute to successful germination and establishment of *S. scoparium* on abandoned gravel spoils in the New Jersey Pinelands. Using a split-split plot, randomized complete block design, I investigated the effects of fertilization, tillage, and mulch depth on germination and growth over multiple growing seasons. I found that plants seeded with 2 cm of mulch germinated in half the time as plants seeded without mulch or with 6 cm of mulch. Furthermore, treatments with 2 cm of mulch had germinating densities almost 10 times greater than those without mulch (104 individuals / m^2 vs. 8 individuals /m^2). Tilling also

influenced early plant growth but had no long term effects on plant size. Fertilization was associated with increases in plant cover over the course of the study but only when coupled with the use of mulch. These results suggest that, on severely degraded soils, mulch depth and tillage influence seedling germination (density), whereas mulch depth and fertilization influence long-term growth. Developing a successful and economical revegetation strategy will play a key role in facilitating the restoration process and will be of utmost value for preserving and protecting highly disturbed pine barrens habitats.

INTRODUCTION

Selecting the native plants or suite of native plants that will best facilitate ecosystem recovery is among the greatest challenges facing restoration ecologists. To be able to develop successful restoration strategies, it is important to first understand the conditions required for a given species to germinate and establish itself on disturbed soils. The ideal species needed to facilitate recovery may not necessarily be one that is dominant or even present in the target habitat (Maestre et al. 2001; Padilla and Pugnaire 2006). Thus, plants comprising target habitats may be ill suited to "pre-restored" conditions (del Moral et al. 2007). Highly disturbed soils can have structural problems, dysfunctional nutrient cycles, and absence of below ground biota that are responsible for mineralization and nutrient cycling, (Bradshaw and Chadwick 1980; Bradshaw 1997; Zink and Allen 1998) all of which can prevent desirable plant establishment. Furthermore, restoration is made more difficult when soils are contaminated with heavy metals or when the hydrological conditions are dissimilar to the surrounding intact landscape (Conrad et al. 2002). Because plants differ in their abilities to tolerate different

biotic and abiotic stressors (Grime et al. 1981; Ehleringer and Sanquist 2006; Silvertown and Tremlett 1989), selecting the best suited colonizers will be extremely important for ameliorating ecosystem degradation and facilitating natural recovery processes. Success in the restoration of ecosystem function relies more on the reestablishment of historical successional pathways than on the mere replanting of native plants. Therefore, identifying appropriate native plant species to use for revegetation and determining the factors that contribute to their successful establishment are vital for initiating the restoration process.

Although data exists on the germination requirements of many plant species, less is known about the germination requirements for species exposed to adverse environmental conditions. That is, most germination studies have been conducted in greenhouse experiments where ideal environmental conditions are maintained. The USDA Natural Resource Conservation Center has an extensive database describing such germination parameters (USDA 2005), however extrapolating these requirements to field conditions can be difficult, especially under varying disturbance regimes. Predicting the success of native plant establishment in a restoration site is even more difficult and many plants used for restoration have been tested only through trial and error (Buckley 1989). Restoration sites can be considered "living laboratories" in that every restoration project provides valuable information regarding the propagation of native plant species not only at varying spatial and temporal scales, but also under a variety of different abiotic and biotic stressors (Sauer 1998). Nonetheless, it is extremely difficult to make general predictions of optimal planting methods in disturbed environments without the ability to replicate restoration events (Clewell and Rieger 1997; Montalvo et al. 2002). At best, the

restoration ecologist can extrapolate optimal planting techniques only to similar habitats with comparable disturbance regimes.

To overcome some of the abiotic and biotic barriers associated with degraded habitats, restoration ecologists make use of multiple planting strategies. These include, but are not limited to, seeding, transplanting, soil ripping, tilling, fertilizing, and mulching. The most common approach for the restoration of abandoned or exhausted surface mines is to establish groundcover by seeding (Bradshaw and Chadwick 1980; Holl and Cairns 1994). Seeding is more cost effective for large areas than planting nursery seedlings or relocating intact flora from adjacent communities; the latter of which is more labor intensive and introduces the additional risks of increased plant stress and mortality (Steven 1991). Seeds can be dispersed by hydroseeding, dry broadcasting or drilling. The best technique for a given site depends on the type of species used, topography of the site, and soil condition (Nelson et al. 1970). For example, Montalvo et al. (2002) demonstrated that in compacted soils with no top soil, hydroseeding resulted in better germination success for smaller seeds, while larger seeds were better established with drilling methods. Similarly, regardless of seed size, dry broadcasting on certain mine spoils can result in increased seed loss from erosion or foraging (Nelson et al. 1970). Comparisons of seeding methodologies are limited in restoration research and there is a clear need for additional work in this area (Clewell and Reiger 1997; Grant et al. 2001).

Soil preparation prior to seeding or planting is an important factor in revegetation success. Soil ripping or tilling is often warranted because sites in need of restoration are often highly compacted. These methods break up the soil surface and improve water percolation and aeration while influencing nutrient availability, organic matter content,

bulk density, and soil grain size (Bradshaw 1997). However, soil improvements made by ripping can also promote weed growth (Kotanen 1997). While some researchers have found these soil preparation treatments to be beneficial (Winkel and Roundy 1991), others have found their impacts negligible (Banerjee et al. 2006). The variability among these studies emphasizes the need to conduct site-specific experiments prior to restoration (Montalvo et al. 2002).

The addition of soil amendments is another approach to improving the success of seedling establishment on degraded sites. Amendments may include fertilizers or organic matter, such as mulch. Mulches can include wood chips, straw, hay, sawdust or other organic or biodegradable materials (e.g. fabrics or cloths). Mulches are used for both their biological and physical benefits; improving growing conditions by increasing moisture retention, moderating soil temperatures, and protecting seeds from wind and erosion (Gonzalez-Sosa et al. 2001; Blanco-Garcıa and Lindig-Cisneros 2005; Chen et al. 2007). Too much mulch, however, can inhibit germination by creating anaerobic conditions or acting as a barrier to light (Winkel and Roundy 1991; Haywood 1999). Mulches can also serve as an important source for organic carbon in the soil (Schoenholtz et al. 1992; Bradshaw 1997; Alpert and Maron 2000) and promote development of microbial communities that remove nitrogen from the soil. This helps maintain a nitrogen-poor environment that potentially favors native species over nitrophilous invasives (Zink and Allen 1998). However, when contaminated with non-native seed, mulches can be a source of undesirable exotic species (Kruse et al. 2004). The quality and quantity of mulch needed for plant growth can vary by species or habitat, contributing to the

difficulty in identifying optimal application techniques (Bradshaw and Chadwick 1980; Fimbel 1992: Rokich et al. 2002).

Site-specific investigations that examine the best practices for land reclamation and ecological restoration are quite valuable especially when they can be applied to ecosystems fragmented by similar disturbances. Such is the case in the New Jersey Pinelands (NJP), where sand and gravel extraction sites have left a significant footprint across the region. Rising and receding sea levels occurring over geologic time resulted in unconsolidated sandy soils across most of southeastern New Jersey, supporting a $192 million dollar sand and gravel mining industry for the state (USGS 2006). Because upland soils of the New Jersey Pinelands are highly acidic and oligotrophic, abandoned borrow pits, estimated at over 500 on the outer coastal plain alone (NJDEP 2010), remain devoid of vegetation for decades. These sites also fragment the landscape, disrupting its ecological integrity by changing natural vegetation dynamics, increasing erosion, or altering the frequency or spread of natural disturbances (e.g. wildfire) (Gibson et al. 1988; Luque et al. 1994; Windisch 1998; Backer et al. 2004)

In 2001, New Jersey's Air National Guard (NJANG) reclaimed a portion of an abandoned gravel pit at the Warren Grove Gunnery Range (WGGR) in south central New Jersey, hoping to reclaim the site for use as a Helicopter Landing Zone (HLZ). Their primary goals were to stabilize the highly eroded soils and produce a low stature vegetative cover that would not impede troop traffic or heavy equipment. *Schizachyrium scoparium* (Little bluestem) was selected as the candidate revegetation species because it had been observed colonizing early successional upland sites in the New Jersey pine barrens and elsewhere (Bien, pers. comm.; West et al. 2003).

Schizachyrium scoparium, is a native C4 perennial bunchgrass that, when planted in early spring, germinates by late spring and continues to grow until the first frost (USDA 2002). Plants can grow up to 91cm on fertile soils, but only to a maximum of 46 cm in poor nutrient soils (USDA 2002). Basal shoots are often bluish, hence the common name. It ranges throughout the United States from Canada to Mexico and is commonly found in dry upland sites in New Jersey. *Schizachyrium scoparium* is considered a dry facultative species (FACU-), meaning it is found less frequently in wetlands (Schmid 2001). It grows in small, dense clumps (caespitose), and vegetative reproduction is mostly by tiller expansion as rhizomes are typically absent. Fire has seasonal effects on its survival. Fall, winter, or spring burns typically result in basal re-sprouting and greater post-fire biomass while summer burns tend to kill plants (Steinberg 2002). These increases in biomass after cool season fires result from greater light penetration, higher soil temperatures, and increased nitrogen availability (Hulbert 1988). Summer burns are typically hotter and destroy basal buds and apical meristems at and below the soil surface (Bragg and Hulbert 1976).

Perennial warm season grasses, such as *S. scoparium* are favored elsewhere in restoration projects because they are drought resistant and adapted to dry, infertile soils (Richardson and Bowers 1987; USDA 2002). Their germination is influenced by fertilization and seedbed preparation. For example, Richardson and Bowers (1987) found that on restored mine beds, the highest densities of *S. scoparium* occurred on treatments where mulch or top soil was used instead of fertilizer. Wedin and Tilman (1996) observed decreases in C4 grass biomass (including *S. scoparium*) along an increasing nitrogen gradient. High levels of nutrients may displace stress-tolerant native grasses and

favor non-native plant species (Buckland and Grime 2000). Carbon amendments, like mulch, may enhance *S. scoparium* establishment by maintaining low levels of nitrogen and retaining moisture (Schoenholtz et al. 1992; Zink and Allen 1998; Averett et al. 2004). Davidson et al. (1990) demonstrated that nitrogen immobilization occurs most readily when organic amendments are left on top of the soil surface when compared to incorporating it. Mulch, fertilizer, and seedbed preparation effects on seeded species in disturbed pine barrens soils have not been thoroughly investigated.

Because the growth response of seeded *S. scoparium* was variable on the restored HLZ, it raised questions regarding the relationship between germination success and the effects of soil amendments, tilling, and fertilizing. The purpose of my study was to investigate this variation and determine the most effective treatment for successful germination of *S. scoparium* on abandoned gravel pits in the New Jersey Pinelands. By examining the effects of organic amendments, as well as the effects of seed bed preparation and nutrient additions, optimal strategies for the revegetation of similar gravel spoils or other mechanically-disturbed sites in the Pine Barrens can be developed. Specifically, the objectives of this study were to:

1) Determine the germination requirements for *S. scoparium*, a candidate restoration species, in disturbed upland sites in the New Jersey Pine Barrens
2) Determine the effects of mulch, fertilization, and tillage on germination and establishment of *S. scoparium* on eroded soils.
3) Develop a restoration model using *S. scoparium* for abandoned gravel pits and other mechanically-disturbed upland Pine Barren habitats.

4) Provide recommendations for reclaiming and managing disturbed sites to natural resource managers at WGGR and the New Jersey Pinelands.

METHODS

1. Site Description and Location:

I conducted this study at the WGGR in Burlington County, New Jersey (Appendix 1). The WGGR is an air to ground conventional and tactical military training facility operated by NJANG. The WGGR occupies 3,804 hectares of upland and lowland pine barren habitats and is located in the core preservation zone of the Pinelands National Reserve on the Outer Atlantic Coastal Plain. Upland pine barren habitats are characterized by dry, sandy, oligotrophic, acidic soils (Woodmansie-Lakehurst Association) (Tedrow 1998). The WGGR supports stands of globally imperiled dwarf pine plains forest (NatureServe 2009). This forest is comprised primarily of the dwarfed Pitch pine (*Pinus rigida*) and the shrub oaks *Quercus marilandica* and *Quecus ilicifolia*. Arborescent pine and oak species are notably absent. In addition, the shrubs are mostly ericaceous and include *Gaylassacia bacchata, G. frondosa*, *Kalmia latifolia* and *Vaccinium palidum*. The ground cover consists of *Carex pennsylvanica*, *Arctostaphylos uva-ursi*, and patchy occurrences of *Corema conradii*, a state endangered species.

The germination field trial was conducted at the Widgeon HLZ, which is located on the western boundary of the WGGR, straddling the Burlington-Ocean County line, New Jersey, USA (39°41'34"N, 74°23'25'W). The HLZ occupies approximately 0.7 hectare of a 2.4 ha abandoned gravel pit formerly comprised of intact pine plains habitat (Figure 22). The sandy substrate is a very coarse Woodmansie soil with very little B

horizon (Figure 23). Operations at the gravel pit began in July 1974 and by August 1976 the site was abandoned (Windisch 1995). During the 1990's, a level portion of the abandoned gravel pit was used for military training until severe erosion prevented its safe use. In order to reclaim the use of the HLZ, the site was restored in spring of 2001. A description of the restoration can be found in Chapter 3. This germination field experiment was conducted on an unrestored portion of the HLZ (Figure 22). Site-specific soil characteristics included low pH, minimal clay, and a high sand fraction (Table 10) Methods for determining these soil properties are described in Chapter 3.

2. Climate:

Climate in the Pine Barrens of New Jersey is seasonally variable (Boyd 2008). Summer months are dominated by the Bermuda High, a clockwise circulation system centered in the Atlantic that brings warm, moist, tropical air from the south. Winter months are dominated by continental air masses that shuttle cold air from the Great Plains and Central Canada (Havens 1998). Mean annual precipitation is 1125 mm, with heavy downpours being frequent in summer (Office of the New Jersey State Climatologist 2008). Summer highs average 23°C, while winter highs average 1°C (Boyd 2008). Differences in annual rainfall and temperature for the study years and the 50 year averages are shown in Figure 24.

3. Experimental Design

To study the factors that influence successful germination of *S. scoparium*, I performed a field experiment using two levels of fertilization (fertilized or unfertilized), two levels of tillage (tilled or untilled), and four levels of mulch application (0cm, 2cm, 4cm, or 6cm). Treatments were arranged in a split-split plot, randomized complete block design (Figure 25). Blocks (n=3) were placed adjacent to each other along the south western side of the abandoned gravel pit. Blocks were separated from one another by two meters and fenced to prevent browsing. Each block was split vertically in half, with each half randomly assigned to a fertilized or unfertilized split-plot. The blocks were also split horizontally and randomly assigned to a tilled or untilled secondary split-plot. Within each split-split plot I installed eight $1m^2$ experimental treatments and randomly assigned each to a seeded or unseeded control with one of four levels of mulch for a total of 32 different treatments per block (Table 11). All $1m^2$ treatments within each block were separated from others by a 0.25 meter buffer.

I applied root-building fertilizer (10-20-10) at a rate of 488 kg/hectare to fertilized treatments and plowed tilled treatments by shovel to a depth of 4 to 5 inches. Experimental treatments were seeded between April 13, 2003 and April 21, 2003 by hand broadcasting *Schizachyrium scoparium* seed (local genotype) at a rate of 22.42 kg/hectare. I then hand spread wood chips to thicknesses of 2 cm, 4 cm, or 6 cm for each mulch treatment. Unseeded treatments were also fertilized, tilled, and/or covered with mulch to serve as controls for studying germination of potential vagrant species. Finally, I covered all treatments with 1-inch wire mesh screen to minimize erosion of the wood chip layer.

4. Plant Monitoring

a) Germination and Survival

I monitored all treatments for germination three times a week from April 14, 2003 until June 30, 2003. Germination time was recorded in days for each treatment. Seedling survival was calculated as a percentage over 6 six months by comparing spring plant density with fall plant density. Individual clumps of *S. scoparium*, a bunchgrass, were considered a single plant. Increases in plant density between spring and fall were reported as 100% survival. Annual survival differences were calculated by comparing 2003 fall treatment mean plant densities with treatment means recorded during fall 2005. Increases in treatment mean plant densities over time were noted but recorded as 100% survival.

For each mulch level, I measured soil temperatures from a random subsample of treatments. Soil temperatures for each mulch treatment were measured at four soil depths (2, 4, 6, and 8 cm) five times between April 14, 2003 and June 30, 2003. Soil temperatures from each mulch treatment level were pooled regardless of whether they were fertilized or tilled. Soil and air temperatures were both measured using a digital temperature probe (Taylor, Model 9841). I measured soil moisture content gravimetrically by removing 10 cm soil cores from all seeded treatment replicates on a single day. Soil cores were bagged, weighed and then oven dried for 48 hours at 70°C. Final dry weights were obtained and treatment means were reported.

b) Plant growth

Plant measures were obtained at the end of the first growing season (October 2003). Density was determined by counting all plants within a treatment. I used the line

intercept transect method to calculate *S. scoparium* cover. Maximum tiller height, measured to the nearest centimeter, was recorded for 10 randomly selected plants. A complete species inventory was conducted for each treatment.

Density, cover and tiller heights were re-measured in October of 2005. At this time I also measured the basal diameters of 10 randomly selected plants for each treatment and recorded the greatest diameter in cm for each individual clump.

5. Data analysis

The effects of fertilizing, tilling, and mulching on germination and growth were analyzed using a split-split plot general linear model (Model III). In this Model III mixed-effect analysis of variance (ANOVA), fertilizing, tilling, and mulch level were the fixed effects while block was the random effect. Distributions were first tested using the Kolmogorov-Smirnov statistic for normality and the Levene statistic for homogeneity of variance. When assumptions for normality or homoscedasticity of variance were violated, data were square root or arcsine transformed. Model III ANOVA's were performed for each growing season. Fisher's protected Least Significant Difference Test (LSD) was used to identify pair-wise differences among treatments.

Because soil temperature data from the fertilized and tilled treatments were pooled, I used a one-way ANOVA to compare differences in mulch treatments for each soil depth. Soil temperature analysis was only performed for 3 of the 6 observations because of missing or limited data at some time points. The Brown-Forsythe test statistic was used when homogeneity of variance was violated. The Brown-Forsythe statistic is a modified F statistic that is more robust for small sample sizes and heteroscedasticity of

variances (Brown and Forsythe 1974). Because preliminary analyses showed no effects of fertilizing and tilling on soil moisture content, I pooled these data and used a one-way ANOVA to test for mean differences in soil moisture among mulch treatments. Differences in germination time among treatments were analyzed non-parametrically using the Kruskal-Wallace test (K-W). Pair-wise differences in germination time for treatments were compared using the Mann-Whitney U Test. For all analyses I accepted statistical significance at $p < 0.05$. The SPSS Statistical Software package (Version 16.0) was used for all statistical analyses.

RESULTS

1. Soil temperatures and moisture content

Sub-surface soil temperatures varied both spatially and temporally and appeared to coincide with seasonal temperature trends (Figure 26). During the germination monitoring period, soil temperature at a depth of 2 cm ranged from 12.5 to 35.6°C. At a depth of 8cm, soil temperature ranged from 12.8 to 30.6°C. Peak soil temperature coincided with peak ambient air temperature on the last day of monitoring day (30-June) in all treatment levels. Lowest soil temperatures where measured in treatments with 6 cm of applied mulch and also corresponded with coolest ambient temperatures (27-May). All three dates analyzed (27-April, 18-June, and 30-June) showed statistically significant differences between soil temperatures and mulch treatments (Table 12). *Post hoc* pair-wise comparisons showed that soil temperatures were consistently higher in treatments where no mulch was added compared to treatments with mulch (Table 13). For all treatments, soil temperatures decreased with increasing soil depth (Figure 27). The

greatest temperature variation was observed in treatments with no added mulch on two of the three sampling dates (27-April and 30-June).

Soil moisture content differed between mulch treatments ($F_{[3,150]} = 5.06$, $p < 0.005$). Unmulched treatments had the lowest moisture content (4.3%), while treatments with 6 cm of mulch had the greatest (5.2%). Individual mean comparison showed that the moisture content of treatments covered with 2 cm of mulch did not differ from other treatments (Figure 28).

2. *Germination onset and survival*

All treatments germinated during the initial 7 week monitoring period. Significant differences were observed among all the treatments (K-W Test, $p < 0.01$) (Figure 29). The earliest germination followed 25 days after seeding in a fertilized-tilled-2 cm mulch treatment. The greatest germination time was 63 days and occurred in an unfertilized-tilled-0 cm treatment. In general, treatments with no mulch had the longest germination times (57 to 64 days), mulched treatments had germination times of 25 to 54 days. Treatments with 6 cm of mulch on average germinated 50.7 days after being seeded. Treatments with 2 to 4 cm of mulch had similar germination times and averaged 29 and 31 days after seeding. Because of the similar germination times between fertilized and tilled treatments, differences in germination times appeared more related to differences in mulch treatment than any other factor.

There were no significant differences in plant survival among treatments during the first six months of the study. Survival between spring and fall of 2003 ranged from 63% in fertilized-untilled-0cm treatments to 100% in unfertilized, tilled or untilled

treatments of varying mulch thicknesses (Table 14, Figure 30). The greatest survival among treatments in 2003 occurred in treatments that were both unfertilized and tilled (three of four mulch treatments had 100% survival at 6 months). In the fertilized treatments only tilled treatments with 6 cm mulch had 100% survival. Annual survival (2003 to 2005) significantly differed among treatments ($p < 0.01$) (Table 15; Figure 30). After two years, survival rates decreased from between 100% and 60% to between 84% and 7% depending on treatment. Differences in survival among treatment were due to differences in mulch thickness ($p < 0.001$) (Table 15). The greatest plant mortality between study years was in unmulched treatments, while mortality between study years was least for mulched treatments. Fertilized treatments tended to have higher rates of survival from 2003 to 2005, but the differences were not statistically significant.

3. Plant Structure

a) Seedling Densities

There were significant differences in seedling densities among experimental treatments in the first growing year ($p < 0.001$) (Table 16). Differences in seedling density were attributed to differences in tilling ($p <0.05$) and mulch levels ($p < 0.001$). There were no differences in two- or three-way interactions among the experimental factors, nor were there any differences between experimental blocks in any of the plant variables. Seedling densities ranged from 106 plants in unfertilized-tilled-2 cm treatments to 6 plants in fertilized-untilled -6 cm treatments (Table 14). Treatments with 2 cm of mulch, regardless of fertilizing or tilling, consistently had higher seedling densities (Figure 31; Table 17). Specifically, unfertilized treatments (tilled or untilled) with 2 cm

of mulch had plant densities significantly greater than all other treatments. There were no differences in plant density among all treatments with either 0 cm or 6 cm of mulch.

Seedling densities in 2005 were less than in 2003 and were significantly different based on treatment factors (Table 18). Density differences were related only to differences in mulch thickness ($p < 0.001$) (Table 18). Unlike 2003, differences in 2005 plant densities were not influenced by tilling ($p=0.974$) and density ranged between 1 and 57 individuals / m^2 (Table 14). Densities in treatments with 2 to 4 cm of mulch were 10 times greater than in treatments with 0 and 6 cm of mulch (Figure 32). Similarly to 2003, unfertilized- untilled- 2 cm treatments had the highest plant density and were the most significantly different from all other treatment combinations (Table 19).

b) Cover

After six months of establishment and growth, plant cover was different among treatments ($p < 0.001$) (Table 16). During 2003, differences in plant cover were influenced by fertilizing ($p < 0.05$), tilling ($p < 0.05$), and mulch ($p < 0.001$) (Table 16). The interaction between fertilizing and mulch was significant ($p < 0.01$). Plant cover ranged from 0 to 22.3% (Table 14). The least amount of grass cover occurred in 0 cm mulch treatments, while greatest cover was in fertilized treatments with 2 cm of mulch (Figure 31).

Although plant density decreased from 2003 to 2005 in all treatments, plant cover increased by 50% in 7 of the 16 treatments (Table 14). After two years of growth, plant cover among treatments ranged between 1.3% and 56.7%. In 2005, treatments continued to show significant differences in grass cover ($p < 0.01$) due to fertilizing ($p < 0.01$) mulch thickness ($p <0.001$), and their interactions with each other ($p < 0.005$) (Table 18).

In 2005, tilling had no effect on plant cover ($p = 0.811$). Like those in 2003, fertilized treatments had significantly higher levels of plant cover than did unfertilized treatments ($p < 0.05$). In 2005, fertilized-untilled- 2 cm treatments had the greatest plant cover when compared to all other treatments (Table 19; Figure 32).

c) Height

In 2003 seedling height, measured as the maximum tiller height per individual of *S. scoparium*, ranged from 1.1 to 7.4 cm with the greatest growth occurring in treatments with 2 cm of mulch (Table 14, Figure 31). There were significant differences in plant height among all treatments ($p < 0.001$) (Table 16). Results of the split-split plot ANOVA model showed that differences in plant height were related to fertilizing ($p < 0.01$) and mulch thickness ($p < 0.001$). Fertilized treatments had significantly taller individuals when compared to unfertilized treatments (Figure 31). In fertilized treatments, there were no differences in plant height among treatments with 2, 4, and 6 cm of mulch. Unfertilized- untilled- 0 cm treatments mulch had the shortest plants and were significantly different from 11 of the 16 other treatment combinations (Table 17).

Mean plant height increased from 2003 to 2005 in 10 of the 16 seeded treatments and ranged between 2 cm and 57 cm (Table 14). The greatest change in plant height (300%) occurred in the unfertilized-tilled- 4cm treatment. Mean height in the four 0 cm treatments remained relatively unchanged over the 2-year period. Plant heights were significantly different among treatments ($p < 0.005$) and were associated with differences in mulch thickness ($p < 0.001$) (Table 18). Two years after seeding, the tallest plants were observed in fertilized treatments with 2, 4, and 6 cm of mulch (Figure 32). Shortest plants

occurred in both fertilized and unfertilized treatments with 0 cm mulch. Pair-wise comparisons of each treatment are shown in Table 19.

d) Basal Diameter

There were significant differences in the basal diameter of individual clumps of *S. scoparium* measured in 2005 ($p < 0.01$). Differences in basal diameters among treatments were associated with differences in mulch thickness ($p < 0.001$) and fertilizing ($p < 0.05$) (Table 19). There were no effects of interaction between the experimental factors. Individuals of *S scoparium* in unmulched treatments were significantly smaller in terms of basal diameter when compared with plants in mulched treatments (Table 19). Two years after seeding, the widest basal diameters occurred in fertilized treatments covered with 2, 4, and 6 cm of mulch (3.5 cm to 3.9 cm) (Figure 32). There were no differences between tilled and untilled treatments on fertilized treatments. The smallest diameters (1.0 cm to 1.5 cm) were measured in unfertilized- 0 cm treatments. In general, unfertilized-untilled treatments of all mulching levels had the smallest basal diameters (Table 19).

6. Naturally recruited species

There were a total of 23 species naturally recruited on to the experimental treatments over the course of this study. All plants that were not seeded but established on treatment plots are listed in Table 20 along with their native status (Schmid 2001). Between 2003 and 2005, native richness increased from 7 to 16 species, while non-native richness decreased from 5 to 4 species. Of the 20 plants inventoried in 2005 only 9 were present in 2003. Conversely, 2 plants observed in 2003 were not present in 2005. With

the exception of one tree species (*Betula populifolia*), all plants inventoried in the unseeded treatments were observed in the seeded treatments. Pitch pine (*P. rigida*) was the most common native plant observed in 2003 and became established on 63% of experimental treatments. Crabgrass, (*Digitaria. sanguinalis*) was the most common non-native species in 2003 and occurred on 81% of experimental treatments. In 2005, *D. sanguinalis* remained the most common non-native plant, while jointweed (*Polygonella articulata*) was the most common native species.

In 2003, 50% of experimental treatments had at least one naturally recruited native plant (Figure 33). By 2005, 87% of experimental treatments had a naturally recruited native plant. In 2003, non-native plants were observed on 94% of experimental treatments, but by 2005 non-natives decreased to 87%. Fertilized and unfertilized treatments with 2 to 4 cm of mulch had the highest numbers of naturally recruited native species.

7. Control treatments

Unseeded treatments (controls) were established to compare the effects of fertilizing, tilling, and mulching with seeded treatments of *S. scoparium*. However, *S. scoparium* germinated on 15 of the 16 unseeded treatments. Establishment of *S. scoparium* on the unseeded controls was likely a result of over-blown, hand broadcasted seed from seeded treatments. Despite its occurrence on unseeded treatments, densities were negligible when compared with those observed in seeded treatments (Figure 34) and results of the split-split plot ANOVA model for the unseeded controls showed no differences among experimental factors ($p=0.08$).

In 2003, three native species were observed on 75% of unseeded treatments (controls) (Table 20). All but one of these (*B. populifolia*) was observed in seeded treatments. Non-native plants were observed on 81% of the control treatments and all five non-native species inventoried were observed on seeded treatments. The unseeded treatments with the highest number of germinating non-natives included the fertilized-tilled- 4cm (richness = 4), fertilized-tilled- 2cm (richness =3), and unfertilized-tilled- 2cm (richness =3) treatments (Figure 34). The greatest number of native plants observered on an unseeded control was two. The most common native species found on control plots was *P. rigida*.

DISCUSSION

The results of this study suggest that plant germination, survival, density, growth, and recruitment on upland gravel spoils in the New Jersey Pinelands are influenced by different soil preparation treatments. These data suggest that the factors essential for germination and first year survival are not the same factors important for long term survival and growth.

1. *Germination*

Seed germination is complex and controlled by several factors. (Raven 1992). The metabolic processes responsible for germination are initiated when optimal temperatures are reached and sufficient moisture has penetrated the seed coat (Salisbury and Ross 1992). Seeds may require specific circadian temperature cycles, an extended period of dormancy, ripening, or scarification in order to germinate (Bewley 1997; Koornneef et al. 2002). If optimal soil and environmental conditions are not achieved, germination may be

delayed, sometimes by decades, and seeds may become less viable as a result of dessication. In this study, the use of mulch strongly influenced soil temperature and moisture. In early spring, mulched plot soil temperatures were correlated with differences in mulch thickness, with unmulched treatments having consistently warmer subsurface temperatures. As seasonal air temperatures increased, mulched plot soil temperatures became more similar despite varied mulch thickness, though unmulched treatments remained comparatively warmer at all soil depths. These findings are consistent with those of previous studies in which the addition of mulch reduced soil temperatures and soil evaporation by protecting the soil surface from solar radiation and moderating soil heat flux (Ross et al. 1985; McGuinnies 1987; Bussière and Cellier 1994; Gonzalez-Sousa et al. 2001).

Germination times varied with mulch thickness. Treatments with 2 or 4 cm of mulch had the shortest germination times. Seedlings emerged between 25 and 35 days at subsurface soil temperatures between 18.0°C and 19.1 °C. In comparison, unmulched treatments and 6 cm treatments had delayed germination. Soil temperatures for unmulched treatments were 22.1°C, and soil temperatures below 6 cm of mulch were 17.4°C. Plants emerged from treatments with 6cm of mulch around day 50, at which point soil temperatures increased to 19°C. Germination in unmulched treatments did not occur until day 63 when soil temperatures stabilized around 20.7°C. Although germination for these unmulched treatments coincided with warmer air temperatures, it should be noted that a cold spell in late May (see Figure 24) reduced soil temperatures in all treatments. The drop in air temperature was thought to lower the higher soil temperatures on bare treatments, improving germination conditions. These germination

temperatures are consistent with data reported for USDA greenhouse studies that have demonstrated that daytime temperatures of 20-30°C are required for germination of *S. scoparium.* Additionally, alternating temperatures of 16.5°C and 27 °C during 24-hour cycles increases germination success, whereas more extreme temperatures inhibit germination (Sabo et al. 1979). Extreme variations in temperature can affect critical soil microbial dynamics (Parker and Larson 1962; Prescott 1990; Pietikåinen 2005). Although germination times differed among treatments, the optimal mulch levels (2-4 cm) appeared to be those with moderate soil temperatures.

Because seeds were dry broadcasted, the limited success of germination in treatments without mulch may have been due in part to other factors. Without a protective mulch layer, broadcast seeds are susceptible to being washed away or depredated by rodents and birds (Nelson et al. 1970; Windsor and Clements 2001). Furthermore, because *S. scoparium* seeds are awned (bristled), mulch may be important for creating stable microsites for germination.

The effect of mulching on soil moisture is less understood. Soil moisture may be related to soil type, mulch type, application rates, rainfall, and topography (aspect and slope), among other characteristics (Hopkins 1954; Deyer and Sencindiver 1985; Norland 2000). The benefits of mulch for soil moisture conservation have been inconsistent across studies (Winkel and Roundy 1991; Chen et al. 2007; Dahiya et al. 2007; McGuinnies 1987; Bergelson 1990; Athy et al. 2006;). Ross et al. (1985) showed that mulch may conserve soil moisture by reflecting solar radiation, lowering soil temperatures, and reducing evaporation, with bare soils more susceptible to soil evaporation. Although the benefits of using mulch have been demonstrated, the benefits of increasing mulch

thicknesses are unclear (Athy et al. 2006; Chen et al. 2007). Thicker mulch layers may impede germination and reduce germination success like that observed on treatments with 6 cm of mulch (Windsor and Clements 2001; Rokich et al. 2002). Because I found no differences in soil moisture content among treatments with varying levels of mulch, the delay in germination and growth in treatments with 6 cm of mulch was most likely due to lower temperatures, physical hindrance, and/or seedling etiolation.

Although the design of this study did not focus on the effects of tilling on soil temperatures and moisture, other studies have reported that tilling can influence soil temperatures by modifying the thermal inertia of compacted soils (Al-Darby and Lowery 1987; Johnson and Lowery 1994; Mahboubi and Lal 1998; Licht and Al-Kaisi 2005). In addition, changes in soil temperatures due to specific tillage techniques are a consequence of increased microbial activity resulting from the mixing of organic material into the soil (Alvarez et al. 1995). Licht and Al-Kaisi (2005) observed that tilling can influence soil moisture content by increasing water infiltration rates. In this study, I found no significant differences in soil temperature or moisture between tilled and untilled treatments, suggesting that differences in temperature and moisture were associated more with mulch thickness than with tilling.

2. Plant survival and density

The period of growth after germination is a critical time for plant survival, as newly emerged seedlings are susceptible to many environmental stressors (Grime et al. 1981). Studies have shown that high mortality rates for seedlings occur during summer months (Silvertown and Dickie 1981; deJong and Klinkhamer 1988). For this study, survival rates among all treatments for the first year did not differ statistically, ranging

from 63% in unmulched treatments to 100% in mulched treatments. The highest plant densities were measured in unfertilized treatments with 2 cm of mulch while the lowest seedling densities were measured in untilled treatments with no mulch and treatments with 6 cm of mulch. The higher seedling densities during the first year were influenced more with mulch depth and tilling than by fertilizer. These data suggest that abiotic conditions, such as soil temperature, are more important for germination than nutrient availability. Therefore, tilling and mulching on pine barren gravel spoils are the most important factors for germination success and first year seedling survival.

Soil moisture is considered one of the most important factors for successful restoration (Schoenholtz et al. 1992; Banerjee et al. 2006). However, the high seedling survival rates among all treatments during the first 6 months of growth in this study may be attributed more to the drought resistant trait of *S. scoparium*. Because of its ability to grow in drought conditions, it is often selected for restoration studies (Smith et al. 1998, Derner et al. 2004). Richardson and Bowers (1987) demonstrated that when seeded on mine spoils without added irrigation or fertilization, *S. scoparium* performed well in terms of germination and survival. In addition, in my study, survival may have been enhanced by the limited competition from other plants.

After three years, plant densities within plots decreased between 16 and 91%. Mortality was greatest in unmulched treatments, while fertilized mulched treatments had the smallest decline in plant densities. High seedling survival with the use of wood mulch has been reported elsewhere (Datillo and Rhoades 2005). Differences in plant densities at the end of the study were only associated with difference in mulch thickness (Table 15). Plant densities in treatments with 2 or 4 cm of mulch were similar to each other and

significantly higher than other mulch treatments irrespective of any other experimental factors.

While mulch depth played a role in determining seedling survival, all treatments had a steady plant decline in subsequent years. Although the greatest mortality occurred in treatments without mulch, there must have been some additional long-term advantage of mulch beyond its positive effect on germination. Whereas the short-term advantages of mulch are related to improving germinating conditions (e.g. optimal temperatures), the long-term advantages may be related to moisture conservation and enhanced nutrient availability.

The addition of organic amendments such as mulch can affect nutrient availability and the soil microbial community (Zink and Allen 1998; Blumenthal et al. 2003). Organic mulch (e.g. wood chips) provides a carbon source for developing the soil faunal community (Alpert and Maron 2000). Microorganisms decompose organic material, resulting in an increase in microbial biomass and immobilization of important nutrients, such as nitrogen and phosphorous. Nutrient cycling then proceeds as the soil community develops and microbial turnover increases. On Appalachian mine spoils, Schoenholtz et al. (1992) found increases in mineralizable N and organic C on wood chip-amended soil three years after application. In this study, early nutrient cycling, as well as moisture conservation in mulched treatments, contributed to plant community sustainability over the three years. Additionally, growth and development of *S. scoparium* was enhanced when mycorrhizal associations developed over the course of the study (see Chapter 5).

3. Plant growth

In this study I evaluated the effects of soil preparation on plant growth by measuring vegetative cover, tiller height, and basal diameter. All treatments, except those without mulch, increased in plant growth over the 3-year study period. There were significant effects of all treatment factors on cover in 2003, including an interaction effect of fertilizing and mulch. Fertilized treatments with intermediate levels of mulch had up to 20 times more plant cover than unfertilized treatments with equivalent mulch thickness. The addition of fertilizer enhanced plant growth by releasing a supply of nutrients for plant growth that was originally deficient in the abandoned gravel pit. In highly disturbed soils, especially mine spoils, fertilizer is commonly the first line of defense in the restoration process, and once native grasses are established, reapplication is typically unnecessary (Bradshaw and Chadwick 1980; Dobson et al. 1997). However, fertilizer use may increase the risk of undesirable invasives (Vinton and Burke 1995; Falk et al. 2006), and its outcome on plant growth may vary by species and ecosystem (Gartner et al 1983; Noyd et al. 1996). Furthermore, fertilizer may have undesireable effects on nutrient cycling and decomposition rates (Thirukkumaran and Parkinson 2000). Because *S. scoparium* is a caespitose grass, plant cover is an index of the plants growth of new tillers and overall increase in basal area. Lateral expansion over time results in fragmentation, which contributes to population maintenance of the bunchgrass (Derner 2004). In this study, plant cover measured in 2003 was enhanced by the interaction of mulch and fertilizer, especially at intermediate mulch levels, because the fertilizer improved the growth of a higher density of plants which germinated under optimal conditions.

Differences in plant cover in 2005 were associated with differences in fertilizing treatments, mulch thickness, and the interaction of mulch and fertilizer. Although plant cover was highest in treatments that were mulched and fertilized, fertilized treatments with 2 or 4 cm of mulch, regardless of tilling, had the greatest amount of plant cover (Table 18). Except for the first year, tilling did not influence plant cover suggesting that the benefits of tilling temporary and more important for early seedling survival. Montalvo et al. (2002) also found the greatest benefit of tilling on plant development occurred within the first year of growth.

Plant height was influenced by differences in fertilizer and mulch thickness. Intermediate levels of mulch with the addition of fertilizer at seeding resulted in taller plants. Unfertilized and untilled treatments with no mulch had the smallest plants. Fertilized treatments with 6 cm of mulch had better plant growth than fertilized unmulched treatments, despite similarities in plant density, suggesting that mulch provides early benefits other than promoting germination success. By 2005, plant height differences were associated with mulch thicknesses and similarities in tiller height were less evident between treatments (Table 18). Increases in height over the 3-year study were evident in all treatments, except those without mulch. The greatest differences between study years occurred in unfertilized treatments with 2 to 4 cm of mulch, where tiller height nearly doubled. This increase in growth did not occur in any other treatment and, whereas significant differences occurred between unfertilized treatments of intermediate mulch levels in 2003, by 2005 these differences were no longer apparent (Tables 7 and 9). Since the increased growth was independent of fertilizer, it follows that seeded plants benefited from the addition of mulch beyond the first year. In agricultural

studies where mulch treatments improved growth rates, researchers attributed improvements to increased soil moisture, more efficient nutrient cycling, and cooler soil temperatures (Lal 1974; Schoenholtz et al. 1992; Downer and Hodel 2001). However, beyond germination benefits, mulch treatments in restoration studies have shown mixed results for long-term growth, emphasizing the need for long-term, site-specific studies (Windsor and Clements 2001: Rokich et al. 2002; Blanco-García and Lindig-Cisneros 2005; Athy et al. 2006).

Plant basal diameters measured at the end of the study showed variation in spatial arrangement of tillers among treatments. Unfertilized-unmulched treatments had the smallest lateral expansion. Differences in basal diameters were associated with differences in mulch application. Comparison of individual treatment means showed no differences between plant diameters with 2, 4, and 6 cm of mulch, suggesting that mulch thickness did not influence plant diameter.

4. Recruitment

One of the most important findings of this study is that natural recruitment of native pine barren species occurred and that all plant forms (trees, shrubs, forbs) were represented. Without intervention, upland gravel spoils typically remain devoid of vegetation for long periods of time. However, this study demonstrated that restoration intervention can be effective for re-establishing vegetation on highly degraded gravel pits.

In 2003, only seven native species (including *S. scoparium*) and five invasive species were inventoried. Native species were found only on mulched treatments. By 2005, native species richness increased to 16 species, while non-native richness

decreased to four species. The most common invasive was *Digitaria sanguinalis* which appeared on all but two treatments. Although the source of the invasive species was unclear, seeds were most likely present in the mulch which was imported from a local recycling center. Although growth and cover of non-natives was not the focus in this study, mulch may have helped prevent the proliferation of non-natives by supporting microbial communities that immobilized excess nitrogen (Alpert and Maron 2000; Blumenthal 2003) or acted as a germination barrier (Haywood 1999).

The most common native species identified on treatments was *Pinus rigida.* This dominant pine barren tree was found in all but three treatments by the end of the study. Pine seeds may have been introduced in the mulch or may have been dispersed from the adjacent forest. Local prescribed burning that occurred during the study most likely contributed to the opening and release of seed from the dwarfed pitch pine's serotinous cones.

Recruitment patterns on experimental treatments in 2003 were unclear, but by 2005 native species were established on all fertilized treatments. On unfertilized treatments, native species richness was greater on tilled treatments. The effect of fertilizing, tilling, and mulching on total plant cover was beyond the scope of this study, but it is likely that these experimental factors have effects on diversity and community dynamics. Restoration practices might benefit from germination studies which examine the establishment dynamics of non-seeded plant species to determine which treatments enhance recruitment of native species.

CONCLUSION

Selecting a candidate species for revegetation is paramount for restoration success. Target ecosystem species may not necessarily be the species of choice, because poor soil structure and dysfunctional nutrient cycles may make site conditions unfavorable for the target plant community. Ameliorating the effects of disturbances, especially in mined soils, is imperative for achieving restoration goals. The selection of an opportunistic native species that can survive disturbed conditions may provide a way of re-establishing successional pathways, and restoring historic ecosystem structure and function. Successful revegetation requires indentifying techniques and strategies that will support germination, survival, and growth of candidate species. In this study, I investigated the factors related to successful establishment of *Schizachyrium scoparium* on abandoned gravel pits in the pine barrens of New Jersey. I found that intermediate levels of mulch (2 to 4 cm thick) applied to the seed bed improved germination success, 3-year survival, tiller and basal growth. Plants grown under 2 to 4 cm of mulch germinated faster and had higher densities than plants grown in any other treatment. Mulch also appeared to influence the recruitment of native species. Fertilizing influenced increased plant cover over the course of the study, but only when coupled with the use of mulch. Tilling was not a major factor in any of the measured outcomes.

The results of this study suggest that successful restoration of highly disturbed soils in the pine barrens is facilitated by seeding *S. scoparium*. The revegetation strategies used in this study demonstrate that once the native grass is established, recovery of later seral species is enhanced. Differences in soil and seed bed preparation on outcomes emphasize the need for site-specific and species-specific remediation.

Table 10. Soil characteristics for germination field trial located on an un-restored portion of the abandoned gravel pit.

pH	4.63 +/- 0.01
Bulk Density (g/cm^3)	0.66 +/- 0.03
Perc Rate (L/min)	0.69 +/- 0.03
Grain Size	
Gravel	14.7% +/- 0.01
Sand	84.8% +/- 0.01
Silt	0.4% +/- 0.0
Fines	0.4% +/- 0.0

Table 11: Summary of experimental treatments for the germination field trial. Treatments were repeated for unseeded controls.

Treatment	Description
UFUT0	Unfertilized + Untilled + 0 cm Mulch
UFUT2	Unfertilized + Untilled + 2 cm Mulch
UFUT4	Unfertilized + Untilled + 4 cm Mulch
UFUT6	Unfertilized + Untilled + 6cm Mulch
FUT0	Fertilized + Untilled + 0 cm Mulch
FUT2	Fertilized + Untilled + 2 cm Mulch
FUT4	Fertilized + Untilled + 4 cm Mulch
FUT6	Fertilized + Untilled + 6cm Mulch
UFT0	Unfertilized + Tilled + 0 cm Mulch
UFT2	Unfertilized + Tilled + 2 cm Mulch
UFT4	Unfertilized + Tilled + 4 cm Mulch
UFT6	Unfertilized + Tilled + 6cm Mulch
FT0	Fertilized + Tilled + 0 cm Mulch
FT2	Fertilized + Tilled + 2 cm Mulch
FT4	Fertilized + Tilled + 4 cm Mulch
FT6	Fertilized + Tilled + 6cm Mulch

Table 12. One way ANOVA table of effect of mulch thickness on sub-surface soil temperatures for each monitoring date.

Date	soil depth	df	F	p
27-Apr-03	2 cm	3,15	74.0	<0.001
	6 cm	3,15	73.6	<0.001
18-Jun-03	2 cm	3,35	44.7	<0.001
	4 cm	3,35	83.9	<0.001
	6 cm	3,35	110.4	<0.001
	8 cm	3,35	107.1	<0.001
30-Jun-03	2 cm	3,12	6.4	<0.05
	4 cm[a]	3,10.7	5.5	<0.05
	6 cm[a]	3,10.7	4.7	<0.05
	8 cm[a]	3,10.2	3.7	<0.05

[a] Denotes use of the Brown-Forsythe modified F statistic

Table 13. Pair-wise comparisons (Least Significant Difference) of mulching treatments at different soil depths for three observation dates (A= 27-April-2003; B= 18-June-2003; C= 30-June-2003).

A

soil depth = 2cm

Mulch Thickness	0cm	2cm	4cm
2cm	<0.001		
4cm	<0.001	<0.05	
6cm	<0.001	<0.001	ns

soil depth = 6cm

Mulch Thickness	0cm	2cm	4cm
2cm	<0.001		
4cm	<0.001	<0.05	
6cm	<0.001	<0.001	ns

B

soil depth = 2cm

	0cm	2cm	4cm
2cm	<0.001		
4cm	<0.001	ns	
6cm	<0.001	<0.05	<0.05

soil depth = 4cm

Mulch Thickness	0cm	2cm	4cm
2cm	<0.001		
4cm	<0.001	ns	
6cm	<0.001	<0.001	<0.05

soil depth = 6cm

	0cm	2cm	4cm
2cm	<0.001		
4cm	<0.001	ns	
6cm	<0.001	<0.001	<0.05

soil depth = 8cm

	0cm	2cm	4cm
2cm	<0.001		
4cm	<0.001	ns	
6cm	<0.001	<0.001	<0.05

Table 13 (continued)

C

soil depth = 2cm

Mulch Thickness	0cm	2cm	4cm
2cm	<0.05	2cm	
4cm	<0.05	ns	4cm
6cm	<0.05	ns	ns

soil depth = 4cm

Mulch Thickness	0cm	2cm	4cm
2cm	<0.05	2cm	
4cm	<0.05	ns	4cm
6cm	<0.05	ns	ns

soil depth = 6cm

Mulch Thickness	0cm	2cm	4cm
2cm	<0.05	2cm	
4cm	<0.05	ns	4cm
6cm	<0.05	ns	ns

soil depth = 8cm

Mulch Thickness	0cm	2cm	4cm
2cm	<0.05	2cm	
4cm	<0.05	ns	4cm
6cm	<0.05	ns	ns

Table 14. Summary data table for all *S. scoparium* plant response measures in study years 2003 and 2005.

	Fertilized								Unfertilized							
	Tilled				Untilled				Tilled				Untilled			
	0cm	2cm	4cm	6cm	0cm	2cm	4cm	6cm	0cm	2cm	4cm	6cm	0cm	2cm	4cm	6cm
2003																
Density (individuals/m^2)	35	84	47	18	13	59	41	6	34	106	45	21	8	104	55	7
% Cover	0.7	22.3	12.3	1.7	0.3	12.0	7.7	0.3	0.7	1.7	2.7	1.0	0.0	2.0	1.7	0.0
Tiller Height (cm)	2.4	7.4	6.3	6.5	2.5	6.7	6.5	5.9	1.6	2.3	2.9	2.6	1.1	3.1	3.0	3.4
Germination Onset (days)	57	25	27	50	59	32	29	51	63	32	35	48	59	27	32	54
% Survival	91.2	86.3	74.9	100	62.9	93.5	92.8	90.4	100	100	86.7	100	77.8	92.4	100	90
2005																
Density (individuals/m^2)	3.3	56.7	39.3	14	1.3	42.3	31	4.7	2.7	22.3	24	5	2	51	32	4
% Cover	0.3	24.0	19.0	6.7	0.0	28.0	18.9	2.0	0.0	4.0	7.0	0.7	0.0	8.0	8.7	0.0
Tiller Height (cm)	2	9.4	9.5	11.7	2.2	11.5	11.3	8.3	1.7	5.4	9.6	3.5	2.1	7.8	8	4.7
Basal Diameter (cm)	1.1	2.7	3.1	3.9	1.5	3.6	3.5	3.6	1	1.9	2.8	1.4	1	1.9	2.1	3.1
% Survival	8.3	68.4	84.3	74.5	8.4	72.2	73.4	65	6.7	24	43.3	45.5	18.4	50.2	61.6	45.5

Table 15. Results of the one-way analysis of variance for survival of *S. scoparium* between fall of 2003 and fall of 2005.

Source of variation	df	Type III Sum of Squares	Mean Square	F	Sig.
Intercept	1	9.99	9.99	43.14	0.02
Block	2	0.46	0.23	4.67	0.18
Fertilizing	1	0.36	0.36	7.26	0.11
Block x Fertilizing	2	0.10	0.05	0.58	0.60
Tilling	1	0.02	0.02	0.25	0.64
Fertilizing x Tilling	1	0.15	0.15	1.71	0.26
Mulch	3	2.24	0.75	21.82	0.00
Fertilizing x Mulch	3	0.29	0.10	2.86	0.06
Tilling x Mulch	3	0.05	0.02	0.48	0.70
Fertilizing x Tilling x Mulch	3	0.01	0.00	0.06	0.98

Table 16. Split-split plot ANOVA on 2003 seedling densities (individuals/m^2), cover (%), tiller height (cm), and basal diameter (cm) of *S. scoparium* planted with varying mulch thicknesses, with or without fertilizing, and with or without soil tilling.

Source of variation	df	Type III Sum of Squares	Mean Square	F	Sig.
Seedling Density					
Intercept	1	87381.33	87381.33	702.80	0.001
Block	2	248.67	124.33	0.49	0.672
Fertilizing	1	1160.33	1160.33	4.55	0.167
Block x Fertilizing	2	510.17	255.08	1.35	0.357
Tilling	1	1704.08	1704.08	8.99	0.040
Fertilizing x Tilling	1	184.08	184.08	0.97	0.380
Mulch	3	40228.83	13409.61	39.57	0.000
Fertilizing x Mulch	3	2453.83	817.94	2.41	0.091
Tilling x Mulch	3	1066.42	355.47	1.05	0.389
Fertilizing x Tilling x Mulch	3	395.75	131.92	0.39	0.762
Cover					
Intercept	1	0.96	0.96	404.77	0.002
Block	2	0.00	0.00	0.52	0.659
Fertilizing	1	0.17	0.17	37.33	0.026
Block x Fertilizing	2	0.01	0.00	1.74	0.286
Tilling	1	0.03	0.03	11.85	0.026
Fertilizing x Tilling	1	0.01	0.01	2.24	0.208
Mulch	3	0.48	0.16	20.14	0.000
Fertilizing x Mulch	3	0.13	0.04	5.25	0.006
Tilling x Mulch	3	0.00	0.00	0.12	0.948
Fertilizing x Tilling x Mulch	3	0.02	0.01	0.63	0.603
Tiller Height					
Intercept	1	13.03	13.03	3445.40	0.000
Block	2	0.01	0.00	0.47	0.682
Fertilizing	1	1.39	1.39	171.22	0.006
Block x Fertilizing	2	0.02	0.01	1.48	0.329
Tilling	1	0.00	0.00	0.04	0.847
Fertilizing x Tilling	1	0.00	0.00	0.11	0.759
Mulch	3	1.17	0.39	25.62	0.000
Fertilizing x Mulch	3	0.03	0.01	0.63	0.606
Tilling x Mulch	3	0.02	0.01	0.43	0.737
Fertilizing x Tilling x Mulch	3	0.06	0.02	1.41	0.267

Table 17. Post hoc pair wise similarity matrix for 2003 plant response variables (A=Density; B= % Plant Cover; C= Tiller Height). "*" denotes significant difference between treatment pair at p <0.05. There are no statistical differences between pairs indicated with shaded cells.

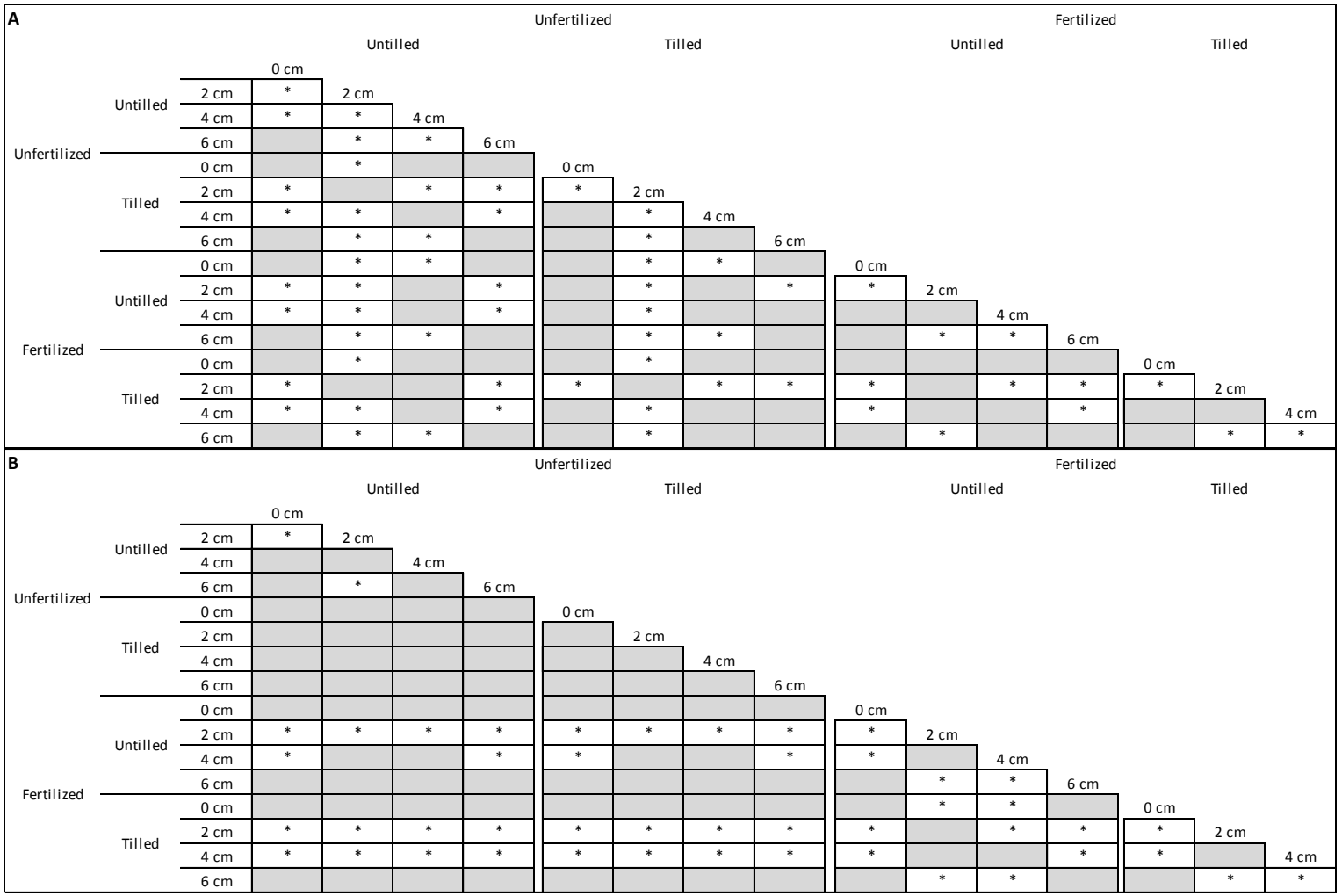


Table 17 (continued)

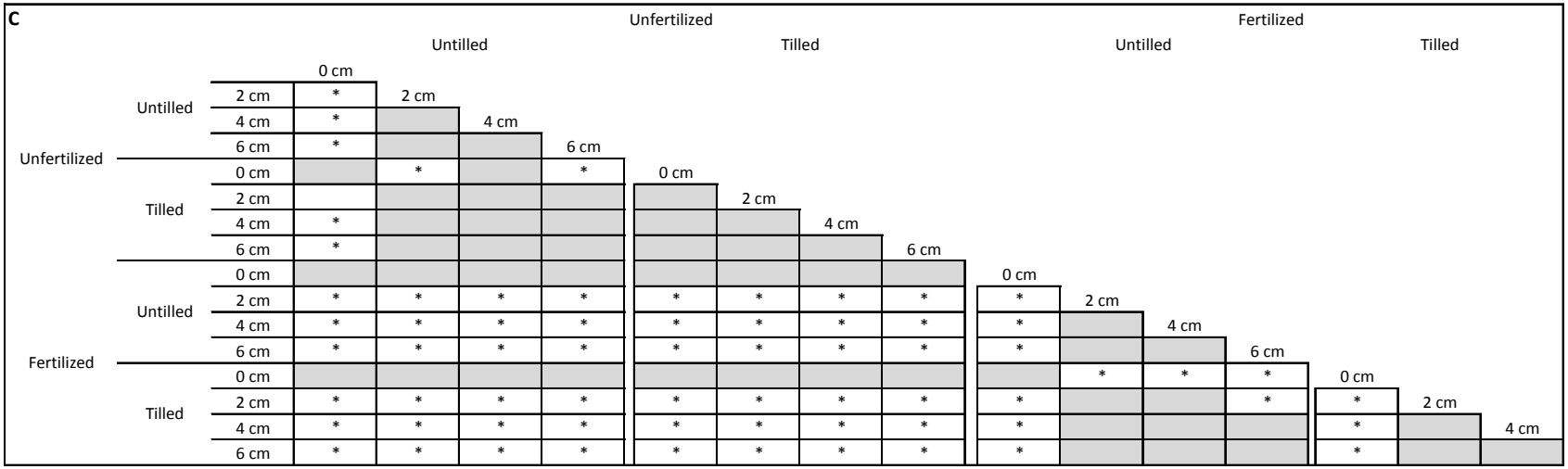


Table 18. Split-split plot ANOVA on 2005 seedling densities (individuals/m^2), cover (%), tiller height (cm), and basal diameter (cm) of *S. scoparium* planted with varying mulch thicknesses, with or without fertilizing, and with or without soil tilling.

Source of variation	df	Type III Sum of Squares	Mean Square	F	Sig.
Seedling Density					
Intercept	1	719.62	719.62	256.72	0.004
Block	2	5.61	2.80	2.94	0.254
Fertilizing	1	6.35	6.35	6.66	0.123
Block x Fertilizing	2	1.91	0.95	0.23	0.805
Tilling	1	0.01	0.01	0.00	0.974
Fertilizing x Tilling	1	9.27	9.27	2.23	0.209
Mulch	3	210.14	70.05	66.67	0.000
Fertilizing x Mulch	3	2.15	0.72	0.68	0.571
Tilling x Mulch	3	4.99	1.66	1.58	0.220
Fertilizing x Tilling x Mulch	3	5.13	1.71	1.63	0.209
Cover					
Intercept	1	0.30	0.30	153.12	0.006
Block	2	0.00	0.00	7.65	0.116
Fertilizing	1	0.09	0.09	359.04	0.003
Block x Fertilizing	2	0.00	0.00	0.07	0.936
Tilling	1	0.00	0.00	0.07	0.811
Fertilizing x Tilling	1	0.00	0.00	0.19	0.682
Mulch	3	0.22	0.07	21.59	0.000
Fertilizing x Mulch	3	0.07	0.02	6.57	0.002
Tilling x Mulch	3	0.01	0.00	0.66	0.587
Fertilizing x Tilling x Mulch	3	0.00	0.00	0.07	0.974
Tiller Height					
Cover					
Intercept	1	2028.27	2028.27	180.08	0.005
Block	2	22.57	11.28	0.81	0.551
Fertilizing	1	84.90	84.90	6.14	0.131
Block x Fertilizing	2	27.71	13.86	4.00	0.110
Tilling	1	2.99	2.99	0.86	0.405
Fertilizing x Tilling	1	2.98	2.98	0.86	0.406
Mulch	3	272.35	90.78	15.05	0.000
Fertilizing x Mulch	3	53.90	17.97	2.98	0.055
Tilling x Mulch	3	17.69	5.90	0.98	0.422
Fertilizing x Tilling x Mulch	3	25.53	8.51	1.41	0.268
Basal Diameter					
Intercept	1	1243.19	1243.19	110.86	0.005
Block	2	29.73	14.87	5.10	0.164
Fertilizing	1	53.18	53.18	18.64	0.013
Block x Fertilizing	2	5.83	2.92	0.20	0.827
Tilling	1	8.54	8.54	0.77	0.422
Fertilizing x Tilling	1	0.40	0.40	0.04	0.858
Mulch	3	66.89	22.30	8.24	0.000
Fertilizing x Mulch	3	13.84	4.61	1.71	0.166
Tilling x Mulch	3	9.30	3.10	1.15	0.331
Fertilizing x Tilling x Mulch	3	19.84	6.61	2.44	0.064

Table 19. Post hoc pair wise similarity matrix for 2005 plant response variables (A=Density; B= % Plant Cover; C= Tiller Height; D=Basal Diameter). "*" denotes significant difference between treatment pair at $p < 0.05$. There are no statistical differences between pairs indicated with shaded cells.

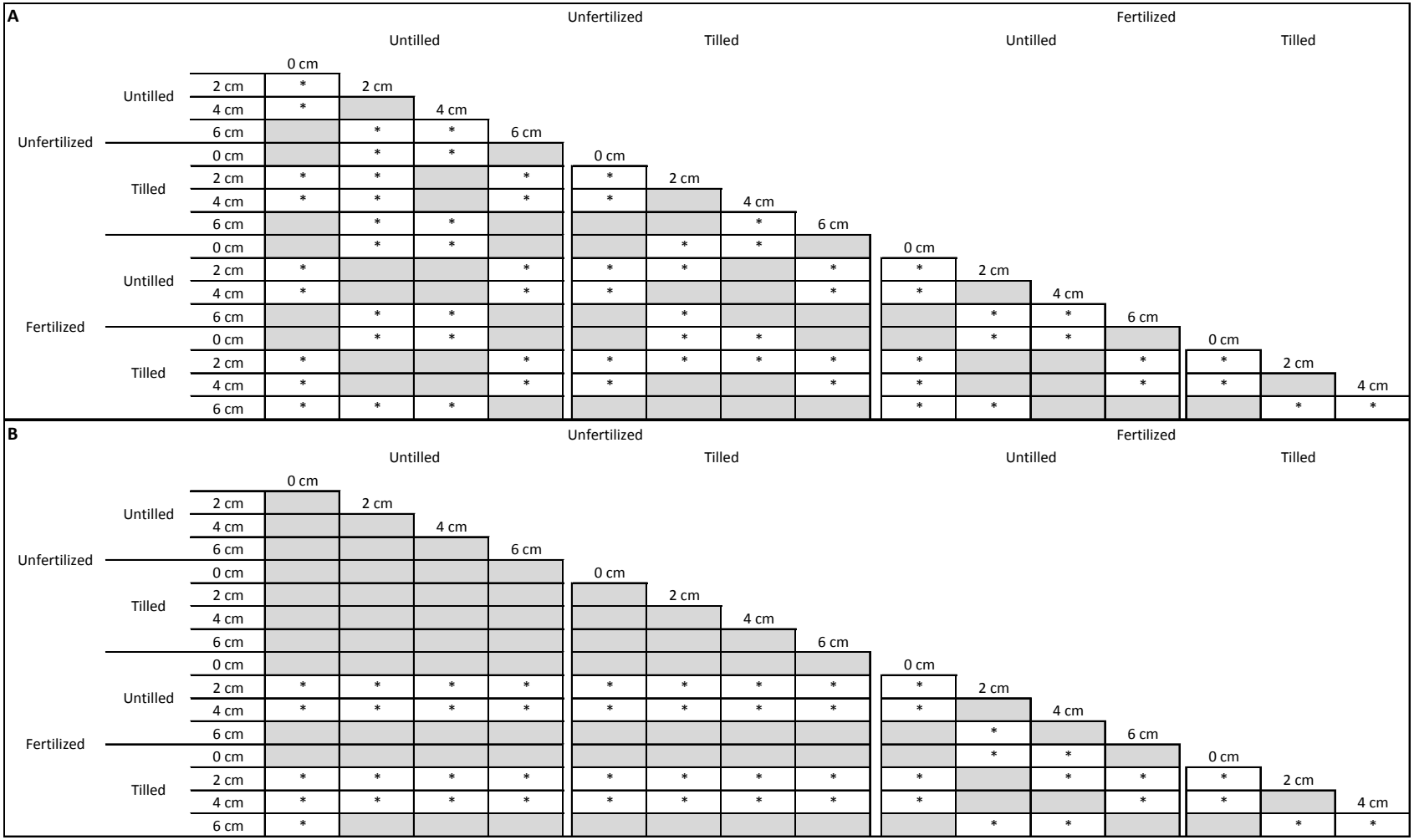


Table 19 (continued)

C

			Unfertilized								Fertilized							
			Untilled				Tilled				Untilled				Tilled			
			0 cm	2 cm	4 cm	6 cm	0 cm	2 cm	4 cm	6 cm	0 cm	2 cm	4 cm	6 cm	0 cm	2 cm	4 cm	
Unfertilized	Untilled	2 cm	*															
		4 cm	*															
		6 cm																
	Tilled	0 cm		*	*													
		2 cm																
		4 cm	*			*	*											
		6 cm		*	*				*									
Fertilized	Untilled	0 cm		*	*				*									
		2 cm	*			*	*	*		*	*							
		4 cm	*			*	*	*		*	*							
		6 cm	*				*			*	*							
	Tilled	0 cm		*	*				*			*	*	*				
		2 cm	*			*	*			*	*				*			
		4 cm	*			*	*			*	*				*			
		6 cm	*				*	*		*	*				*			

D

			Unfertilized								Fertilized							
			Untilled				Tilled				Untilled				Tilled			
			0 cm	2 cm	4 cm	6 cm	0 cm	2 cm	4 cm	6 cm	0 cm	2 cm	4 cm	6 cm	0 cm	2 cm	4 cm	
Unfertilized	Untilled	2 cm																
		4 cm	*															
		6 cm																
	Tilled	0 cm			*													
		2 cm																
		4 cm	*				*											
		6 cm																
Fertilized	Untilled	0 cm																
		2 cm	*	*			*	*		*								
		4 cm	*	*			*			*								
		6 cm	*				*											
	Tilled	0 cm							*			*	*	*				
		2 cm	*				*								*			
		4 cm	*				*			*					*			
		6 cm	*				*			*					*			

Table 20. Inventory of naturally colonizing (unseeded) species for each study year. Presence or absence of plant species was conducted at multiple times throughout the growing season. Control plots are those where *S. scoparium* seed was not sown.

Scientific Name	2003	2005	2003 Control
Natives			
Aristida dichotoma	x	x	x
Andropogon glomeratus		x	
Betula populifolia			x
Carex pennsylvanica	x	x	
Dicanthelium sp.	x	x	
Diodia teres		x	
Euphorbia ipecacuanhae		x	
Hudsonia ericoides		x	
Hypericum gentianoides		x	
Ilex opaca		x	
Krigia virginica		x	
Morella pennsylvanica	x	x	
Panicum sp.	x	x	
Pinus rigida	x	x	x
Polygonella articulata		x	
Sorghastrum nutans		x	
Vaccinium palidum		x	
Introduced			
Digitaria sanguinalis	x	x	x
Oenothera laciniata	x		x
Oxalis sp.	x		x
Polygonum punctatum	x	x	x
Setaria sp.	x	x	x
Taraxacum officinale		x	

Figure 22: Aerial view of the abandoned gravel pit. (1) The helicopter landing zone (HLZ) restored in 2001 by seeding with warm season grasses (*S. scoparium*). (2) The unrestored portion of the gravel pit used as the pre-restored baseline for the functional study in Chapter 5. (3) The portion of the abandoned gravel pit restored in 1997 with pine tree seedlings and evaluated in Chapter 2. (4) The site of the *S. scoparium* germination study described in this chapter.

Figure 23. Exposed soil fraction remaining on the abandoned gravel pit prior to restoration consisting of course to sandy Woodmansie soil with very little A-Horizon remaining.

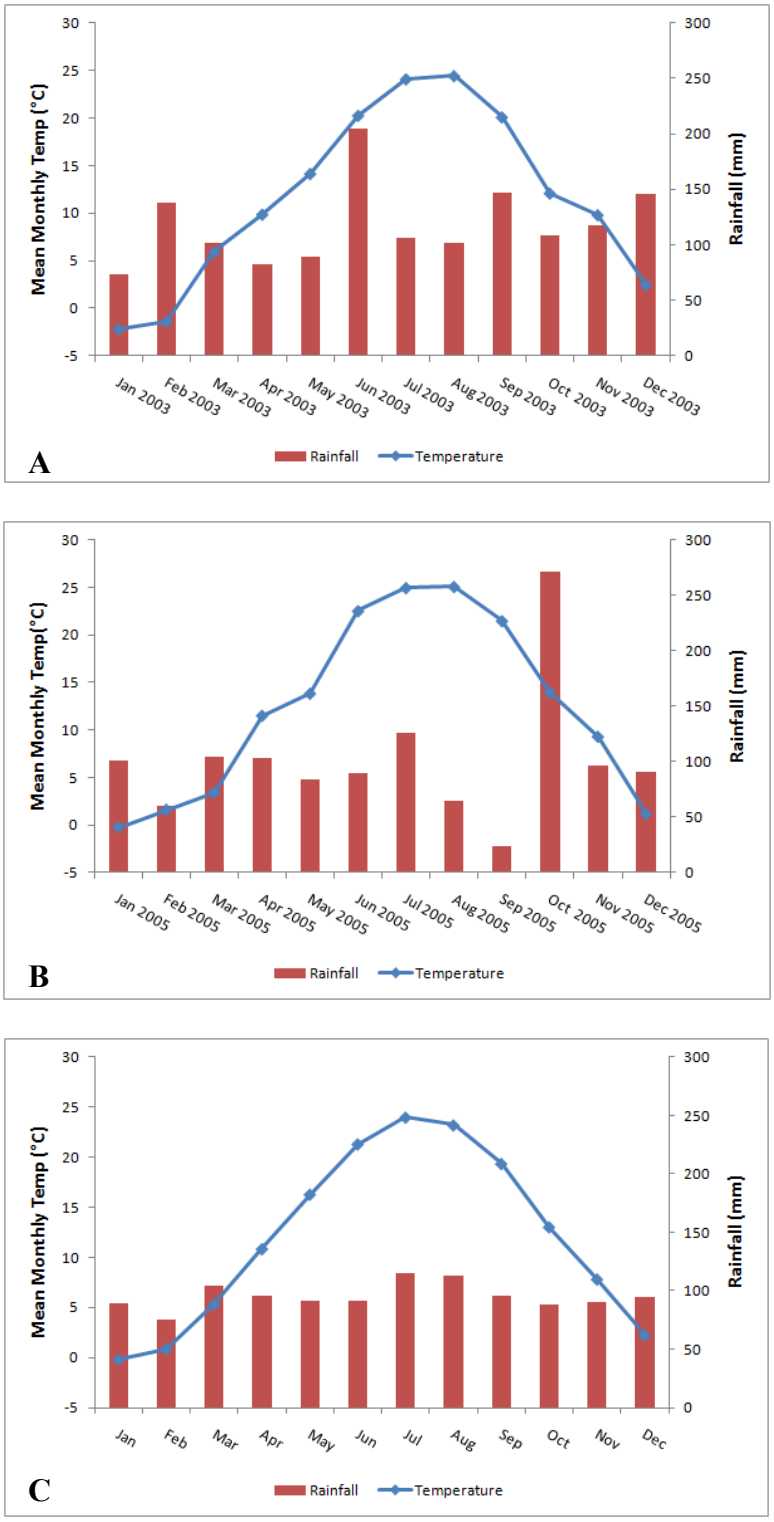


Figure 24. Mean monthly air temperature and rainfall data for study years 2003 (A), 2005 (B), and (C) 50 year average. Annual and averaged derived from mean of 10 weather stations in Central New Jersey.

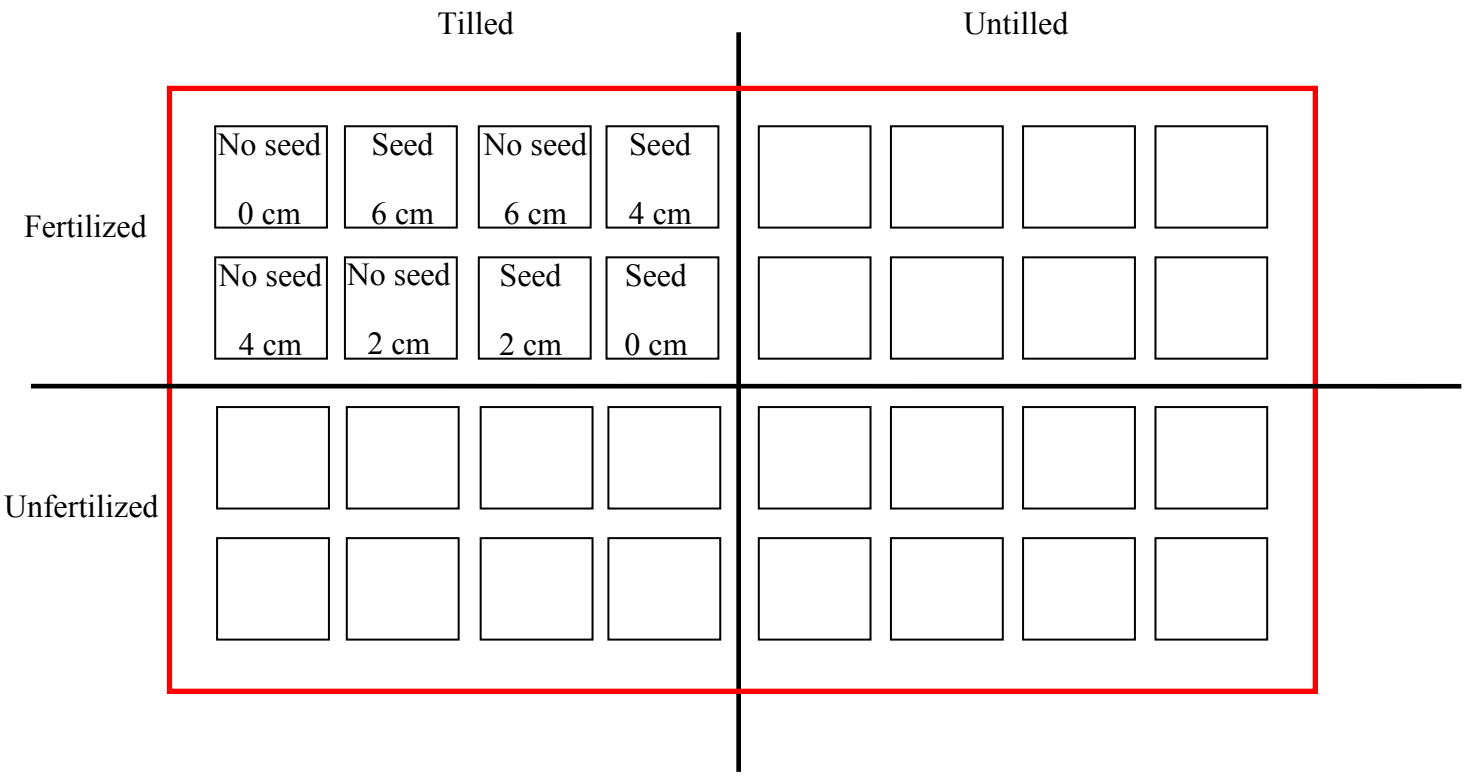


Figure 25. Schematic diagram of the split-split plot randomized block design employed in germination study. Each block (outlined in red and approximately 11m x 6m) was replicated 3 times and positioned adjacent to each other on a level portion of the abandoned gravel pit. Blocks were split horizontally and randomly assigned a fertilization level and split vertically and randomly assigned a tillage level. Experimental $1m^2$ treatments (seeded and non-seeded) were randomly assigned a mulch level for each split-plot. Treatments were separated by 0.5m.

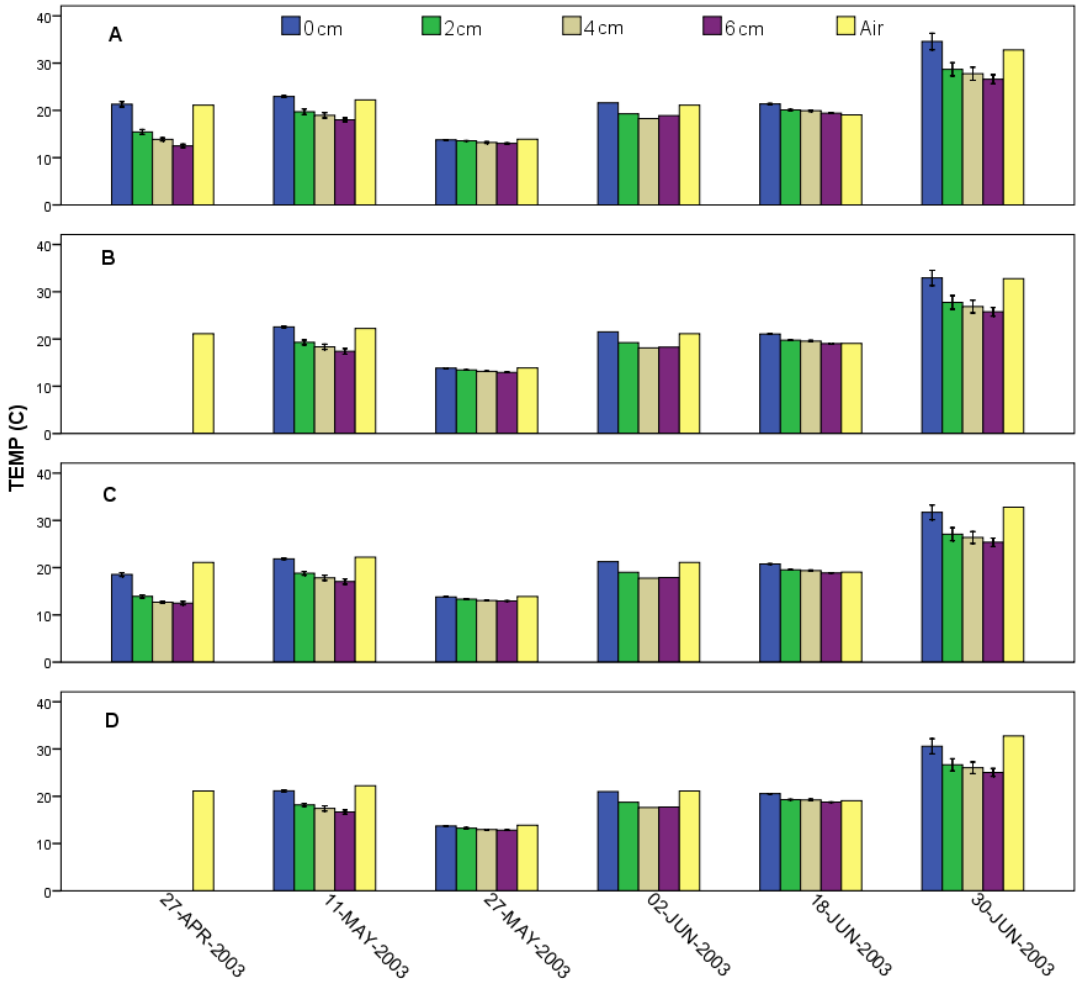


Figure 26. Mean soil temperature and ambient air temperature (°C) for four mulch treatments (colored bars) at four different soil depths. (Panel A= 2cm, Panel B = 4cm, Panel C = 6cm, and Panel D = 8cm) for first 75 days of growth. Ambient air temperature is represented in each panel as yellow bar. Error bars represent +/- 1 S.E.M.

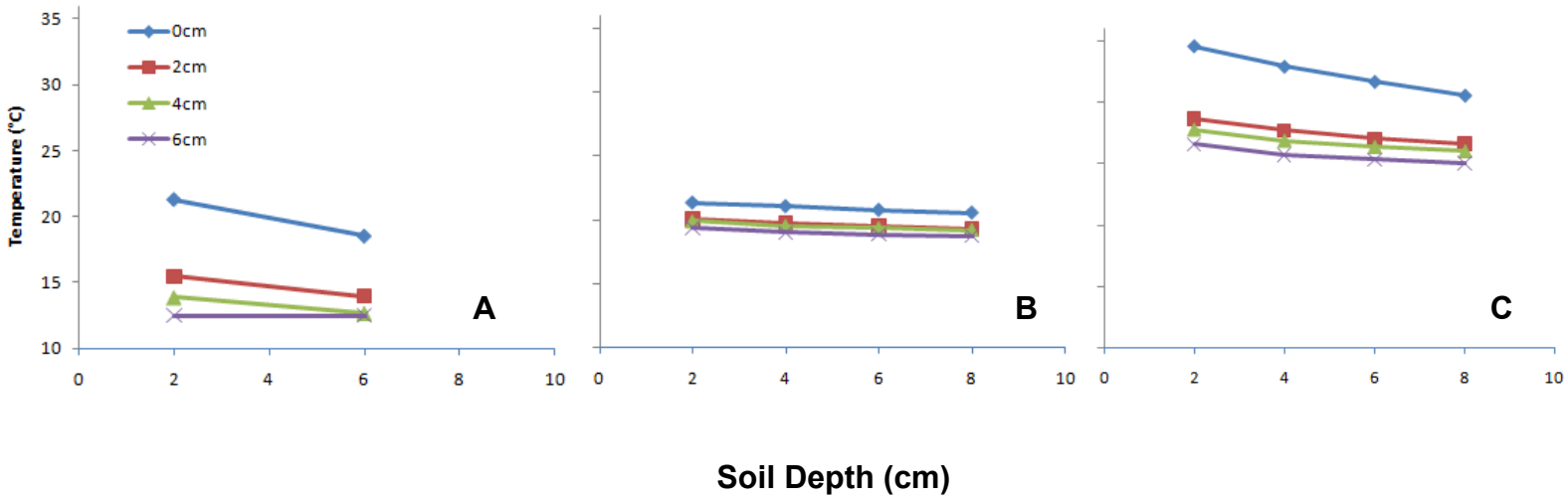


Figure 27. Soil temperature profiles for pooled mulch treatment levels for three monitoring dates (A= 27-April-2003; B= 18-June-2003; C= 30-June-2003).

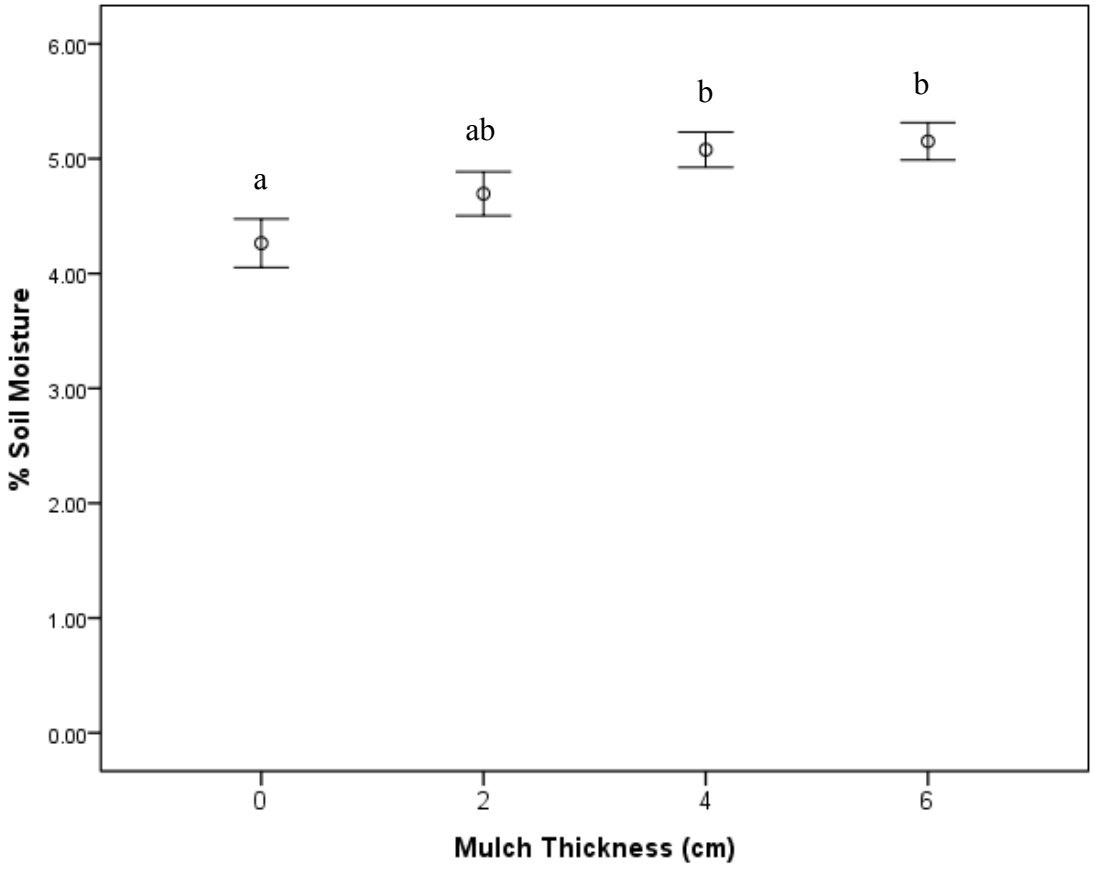


Figure 28. Mean moisture content of soil for different mulch treatments determined gravimetrically from randomly pulled soil cores during summer of 2005. Data points with different letters are significantly different at $p < 0.05$. Error bars represent +/- 1 S.E.M.

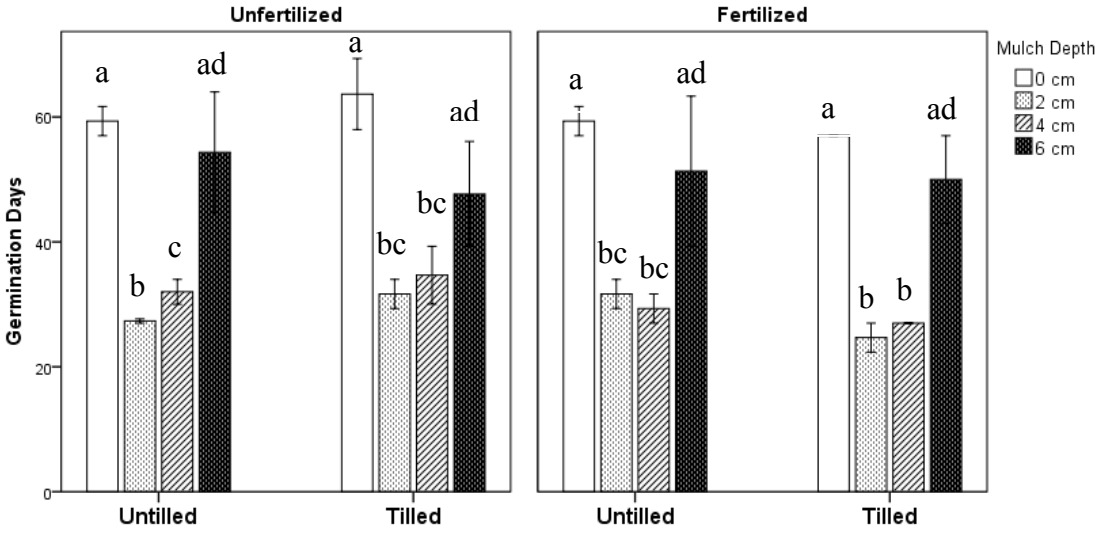


Figure 29. Mean days for germination onset. Data analyzed non-parametrically with Kruskal–Wallace test statistic. Pair-wise post-hoc analyses performed with Mann Whitney U Test. Bars with different letters denote statistical differences at $p < 0.05$. Error bars represent +/-1 S.E.M.

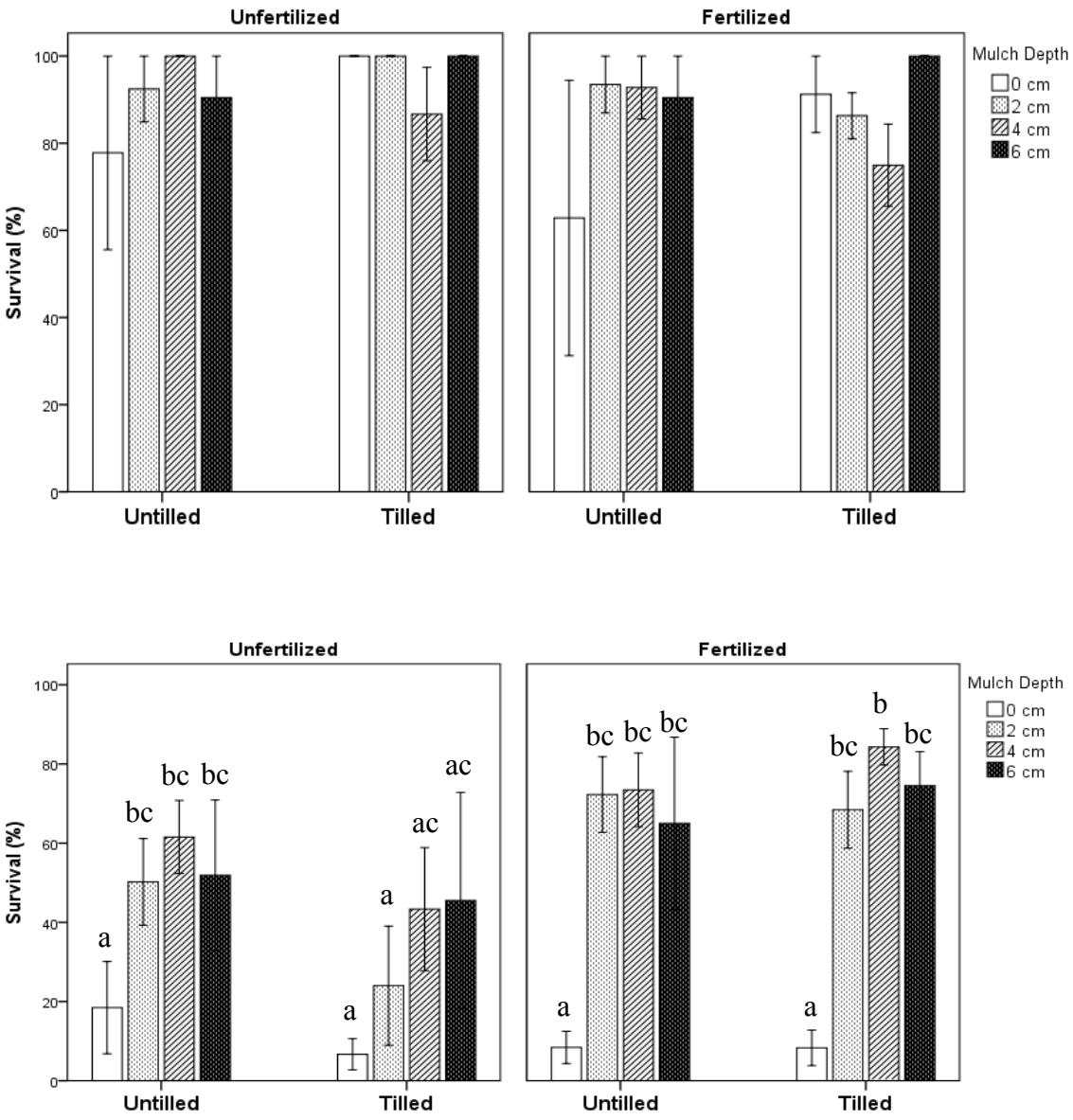


Figure 30. Six month mean survival rate (%) during 2003 (top) and inter-annual survival (2003 vs. 2005) (bottom) for all experimental treatments. Bars with different letters denote statistical differences at $p < 0.05$. Error bars represent +/-1 S.E.M.

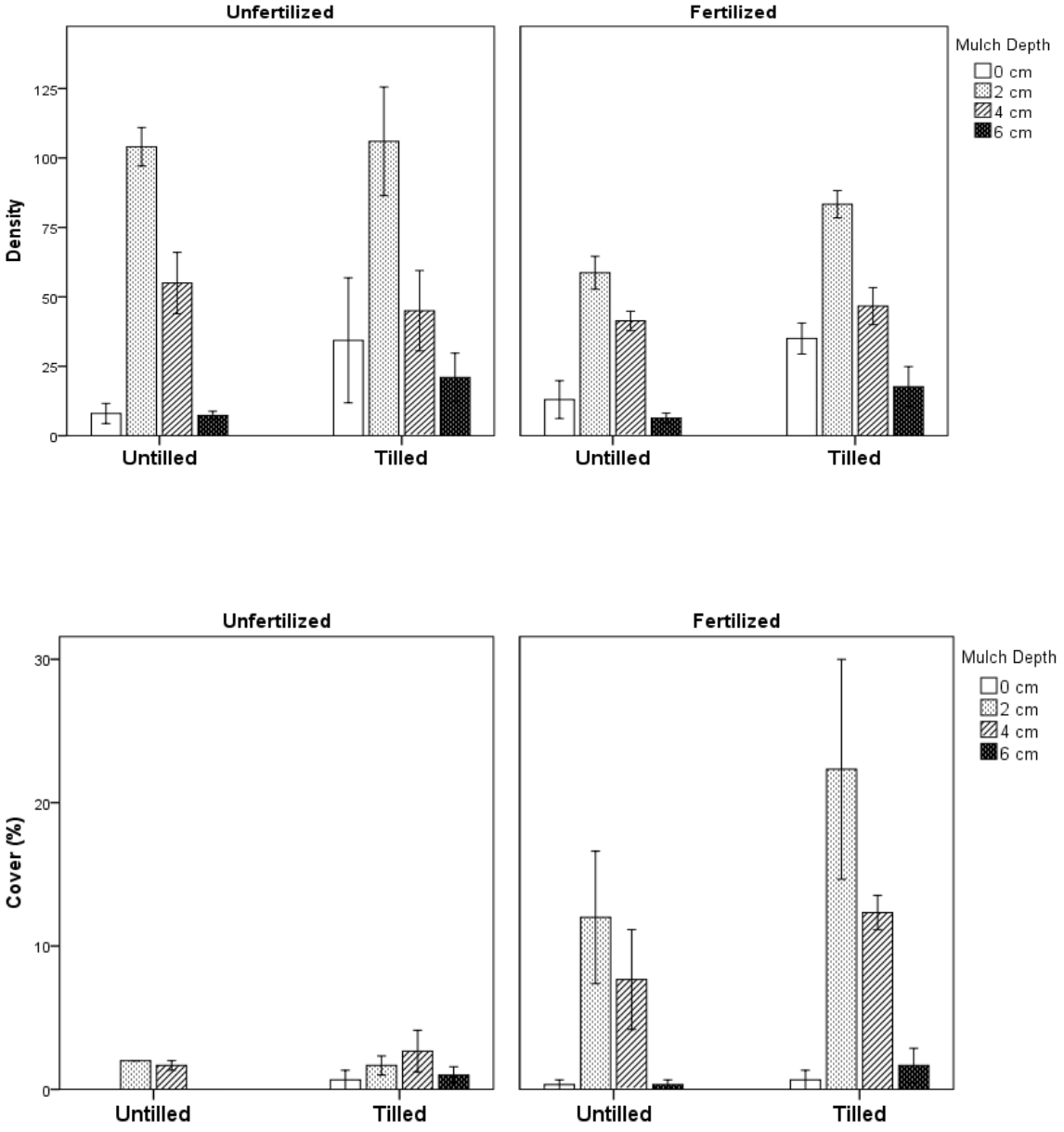


Figure 31. Data for 2003 plant density (individuals/m^2), plant cover (%), and mean tiller height (cm) of *S. scoparium* for all treatments. Pair wise comparisons of means are shown in Table 17. Error bars represent +/- 1 S.E.M..

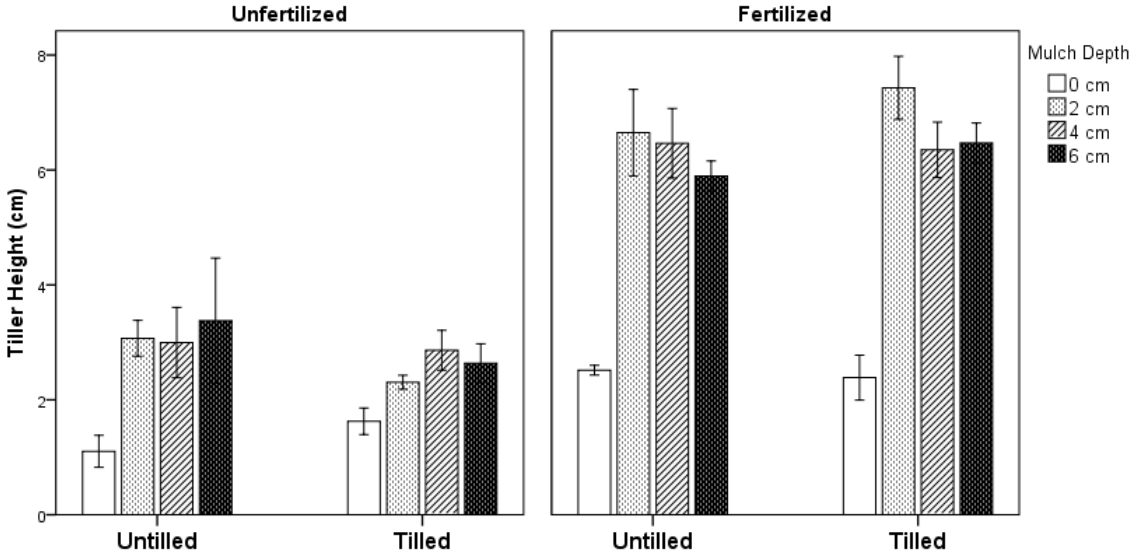


Figure 31 (continued)

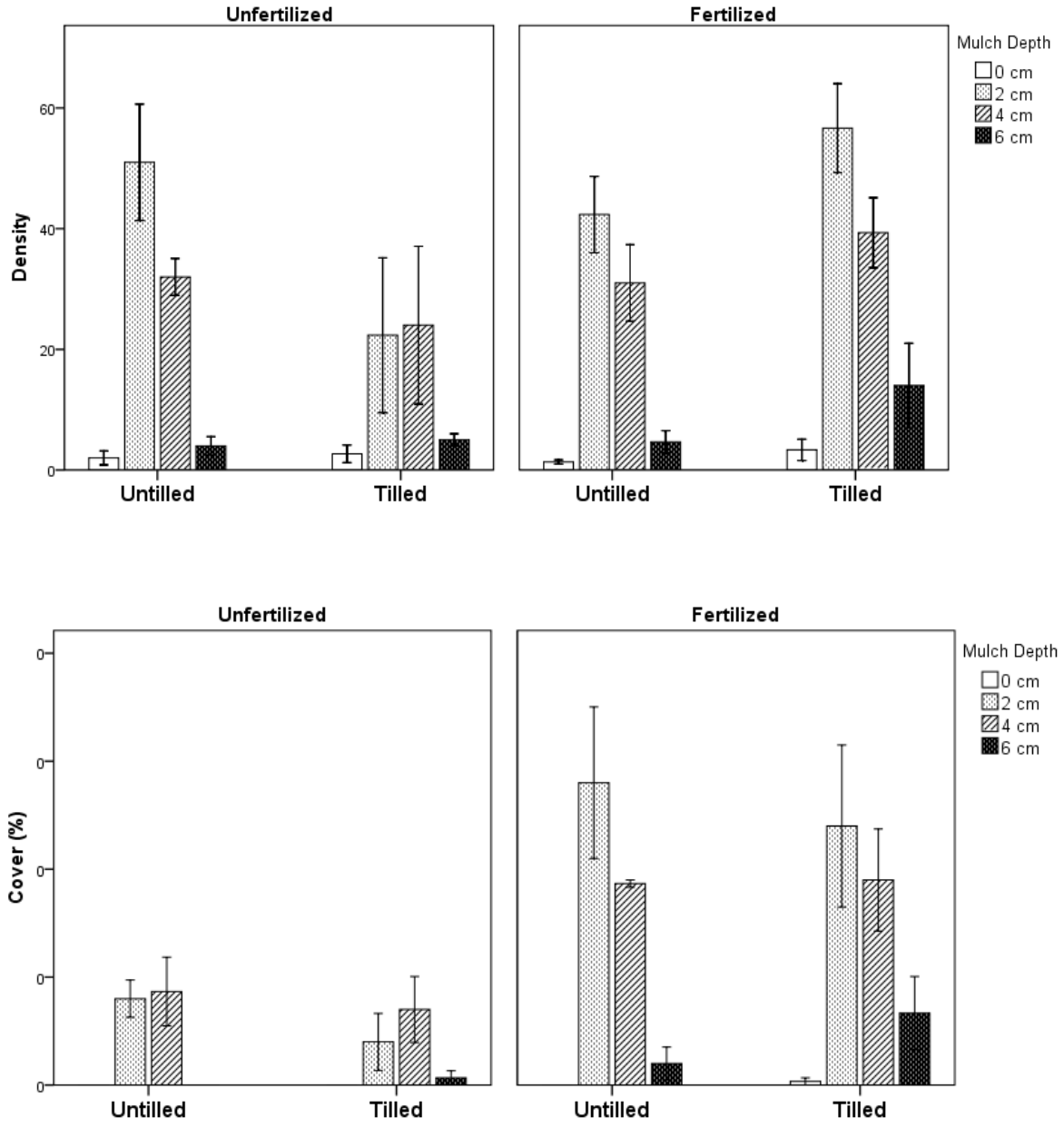
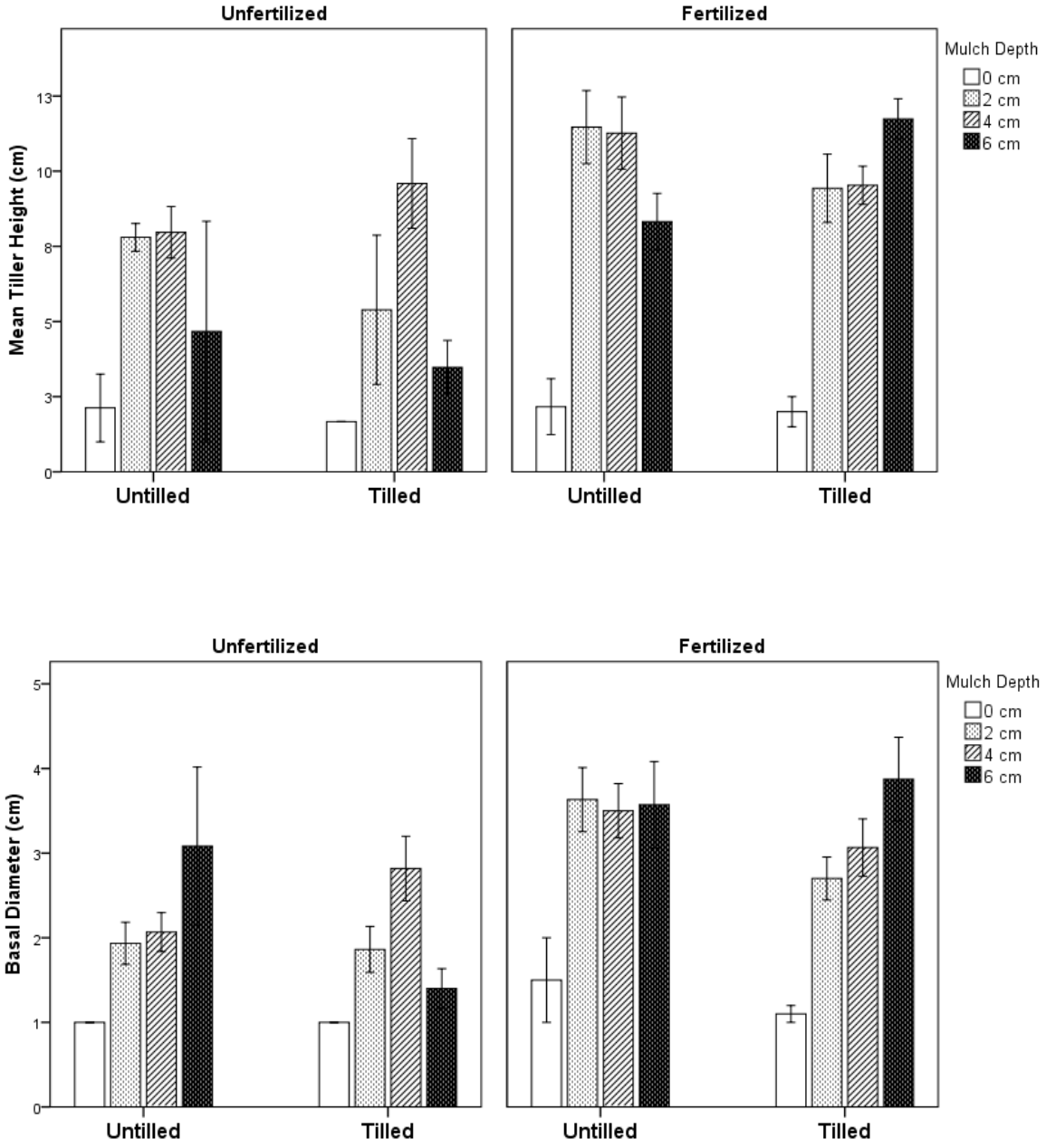


Figure 32. Plant density (individuals/m^2), plant cover (%), mean tiller height (cm) and mean basal diameter (cm) of *S. scoparium* for all experimental treatments in 2005. Pair wise comparisons of means are shown in Table 19. Error bars represent +/- 1 S.E.M.



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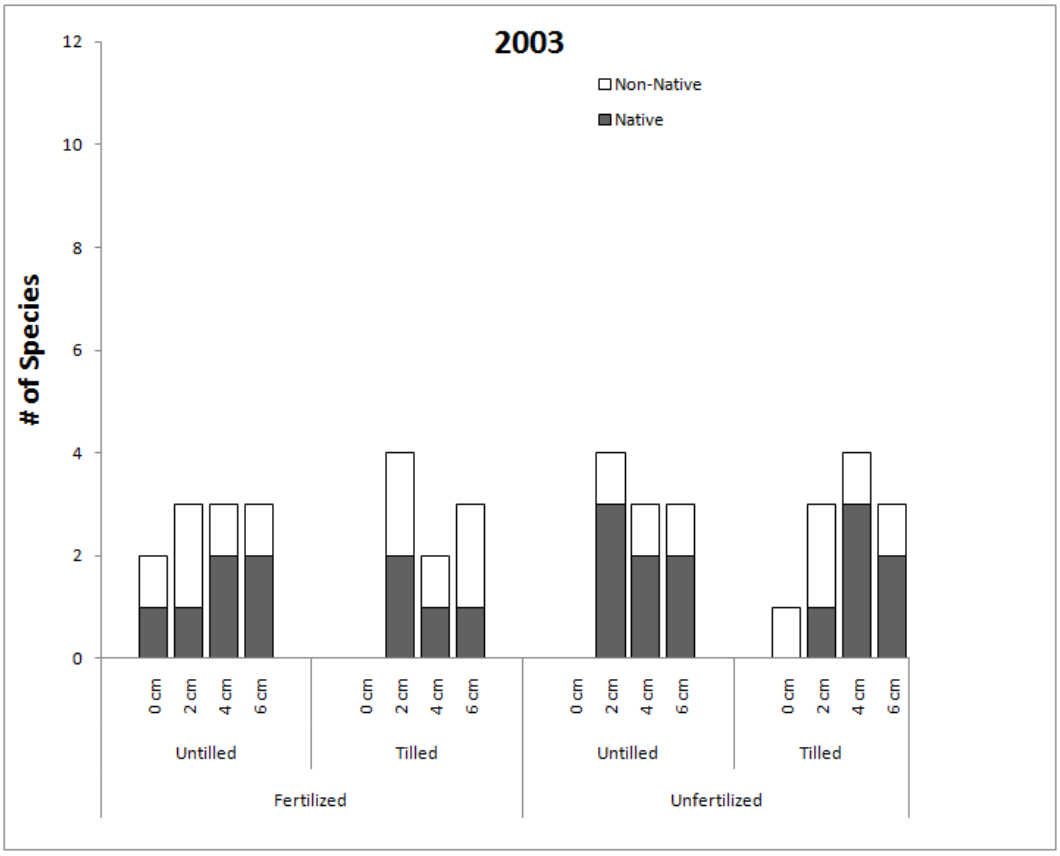

Figure 33. Species richness for the experimental treatments in 2003 (top) and 2005 (middle) and for the control treatments in 2003 (bottom).

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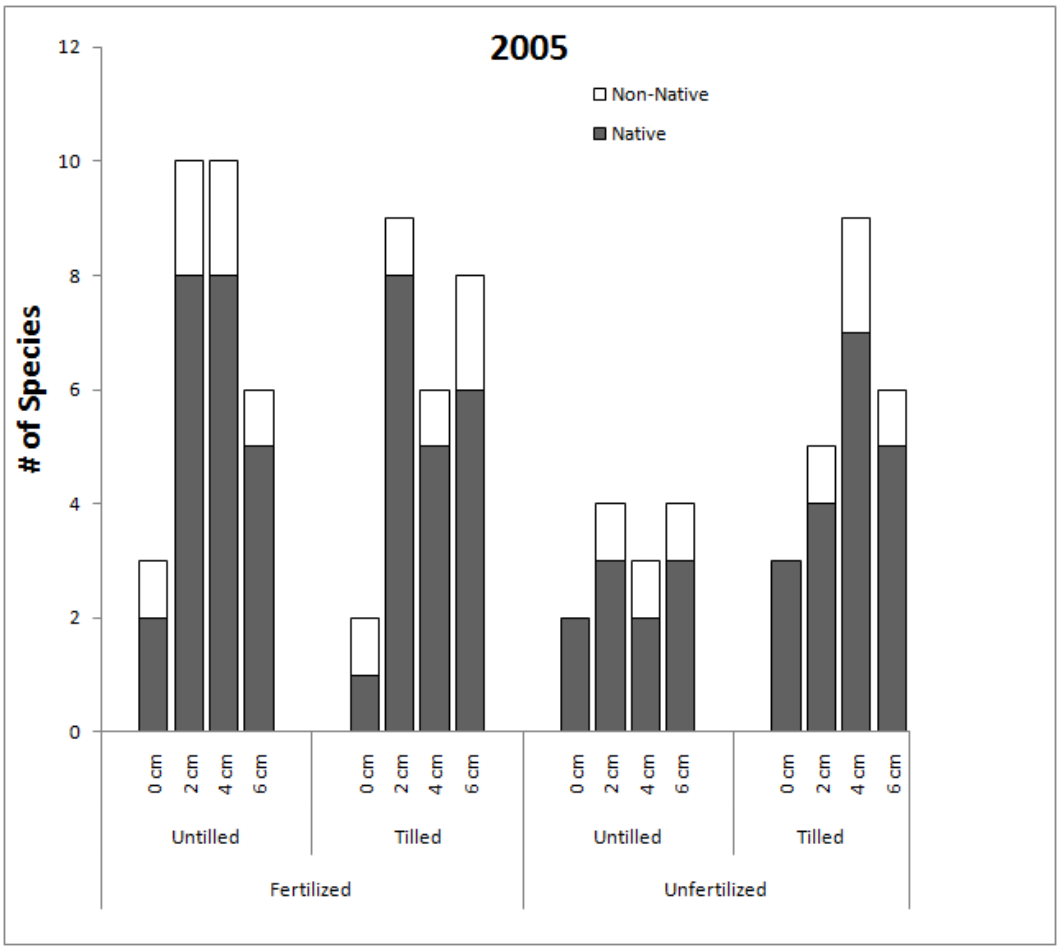

Figure 33. (continued)

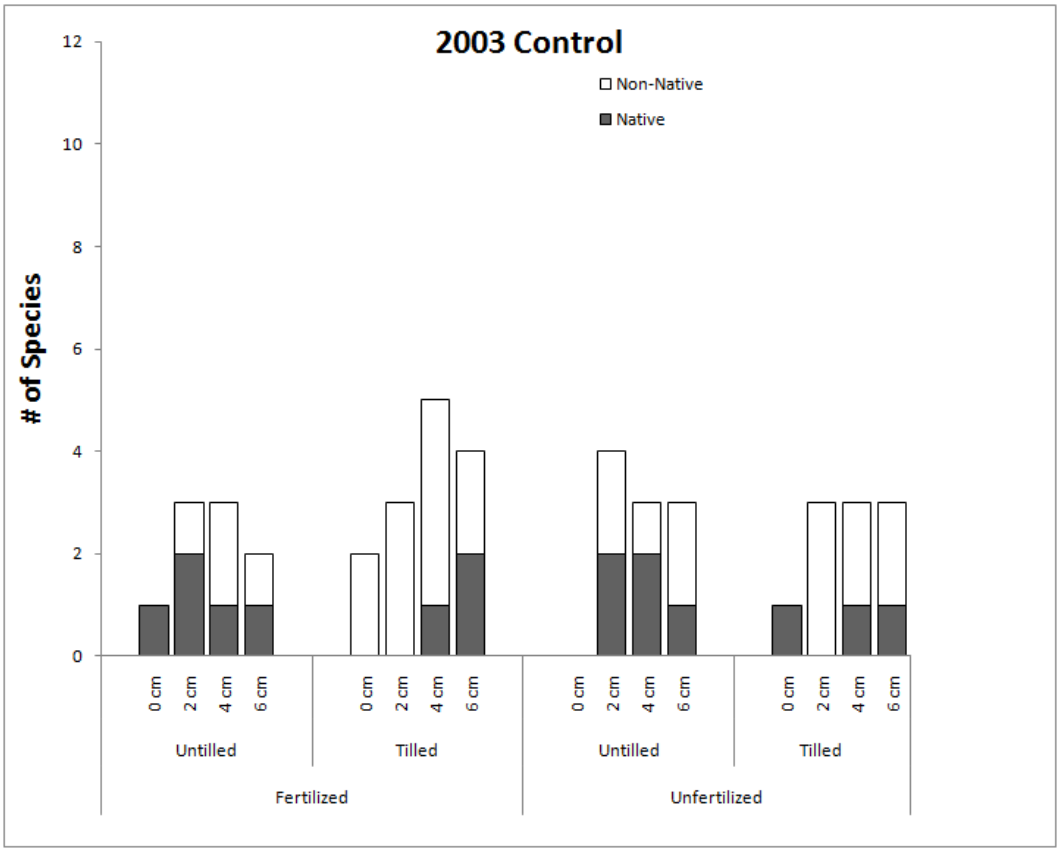

Figure 33. (continued)

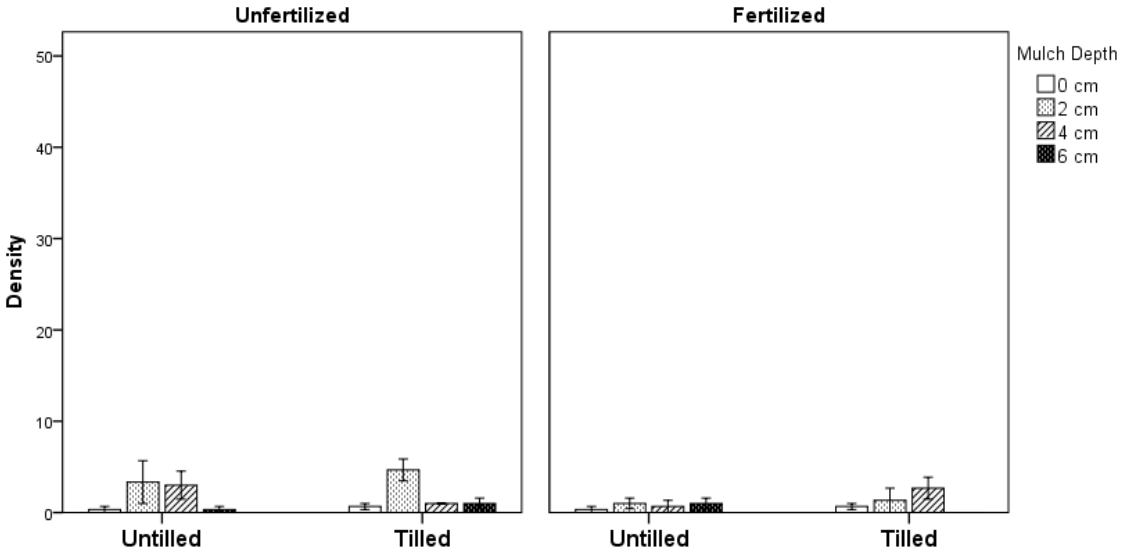

Figure 34. Density (individuals per m2) of *S. scoparium* on the unseeded control plots measured in fall of 2003. Error bars represent +/- 1 S.E.M.

CHAPTER 5: A comparative analysis of below ground function between a restored gravel pit and naturally recovering disturbed pine plains community in the New Jersey Pine Barrens

ABSTRACT

Ecosystem function and structure are driven by biogeochemical processes occurring above and below the soil surface. Understanding the links between the two is important for understanding how plant communities develop or respond to disturbance. Ecological restoration focuses on methods to restore these natural processes and re-establish successional trajectories towards a self-sustaining ecosystem. After first mitigating the disturbance, restoring plant communities involves replacing native plant species or removing undesirable ones. Measuring the efficacy of those steps in terms of structural and functional recovery is paramount to determining restoration success. In this study, I evaluated the practice of using native warm season grasses to restore an abandoned gravel pit in the pine plains of the New Jersey Pinelands, by comparing nitrogen allocation and root development on a restored site with a naturally recovering reference site. I found that planting a stress-tolerant native colonizer, (*Schizachyrium scoparium*) catalyzed natural succession in this oligotrophic environment. Individuals of the warm season native grass had similar foliar C:N ratios and mycorrhizal colonization rates, but different root to shoot ratios. The restored site had significantly higher soil pH, while the reference site had a higher clay fraction and higher root length density. Nitrogen levels varied by season and by site, but both sites displayed similar nitrogen availability patterns. While differences in recovery age between the two sites may have been the most important factor in explaining functional differences, these data suggest

that planting warm season grasses ameliorated site conditions enough to initiate natural recovery towards a self-sustaining pine plains community.

INTRODUCTION

In terrestrial ecosystems, the interactions between above ground and below ground processes have long been appreciated as factors driving both ecosystem structure and function (Grime 2001; Tilman 1982). The soil biota is responsible for decomposition, nutrient cycling, and ecosystem engineering, while the plant community provides the primary source of carbon, influences litter quality and alters microclimates and microhabitats (Wardle 2002). These functions are moderated by environmental factors such as climate, topography and hydrological regime (Haselwandter et al. 1983; Chambers et al. 1987; Dharmakeerthi et al. 2005). Together these processes form complex positive and negative biochemical feedbacks between the soil and plant community.

Despite these linkages, ecologists often partition these two systems, emphasizing research on either one component or the other (Wardle et al. 2004; Bardgett et al. 2005). Given the influence of above ground structure on below ground function, and vice versa, an integral understanding of these two subsystems is vital, however obtaining soil productivity measures can be difficult and costly (Allen 1992). This leads many restoration ecologists to concentrate efforts on above ground structural characteristics and limit attention on soil processes (Ruiz-Jaen and Aide 2005a; Ruiz-Jaen and Aide 2005b). Furthermore, because soil functions tend to recover more slowly than above ground structure, restoration ecologists focus on short term indicators of community recovery

such as vegetation development (Chambers et al. 1994; Kindscher &Tieszen 1998; Morgan & Short 2002). In such studies, structural attributes of restored landscapes (e.g. growth, cover), are used as surrogates of ecosystem function. Harris (2003) notes that without long term monitoring, recently restored sites will likely resemble target ecosystems merely because disturbed sites are typically revegetated with desirable species.

Similarly, forests that are restored by thinning or burning will produce a desirable structure shortly after treatment. This may lead to false estimates of longer term restoration success. Therefore, when long term monitoring is not possible, a better picture of ecosystem recovery may be obtained by supplementing structural measures with functional measures. Measuring ecosystem processes, such as those occurring in soil, can provide information on biochemical cycles and nutrient cycling necessary for long-term stability of the ecosystem (Herrick 2000; Ruiz-Jaen and Aide 2005a). Ecologists may be better able to interpret successional trajectories and quantify ecosystem recovery and restoration success by using comparative analyses of these measures with those of a reference site.

In surface mined landscapes, where vegetation and top soil have been removed, nutrient cycles are dysfunctional or altogether absent (Bradshaw and Chadwick 1980). Restoring the biochemical pathways between soil and plants is the first major step needed for successfully restoring ecosystem structure and function in these landscapes (Bradshaw 1982). Without repairing soil function, these lands can remain in a derelict condition indefinitely. This is because mined soils are depleted of the nutrients necessary for the development of a diverse micro- or macrobiotic community. In the absence of soil

biota, nutrient recycling and mobilization is inhibited and therefore plant growth is limited. Because plant growth is limited, there is minimal accumulation of organic matter, the carbon source for microbes, either at the soil surface or in the rhizosphere (Post and Kwon 2000; Akala and Lal 2001). The absence of soil organic matter in mined soils may be the main factor limiting the development of the soil microbial community and subsequent return of soil processes (Frouz and Novakova 2005; Sourkova 2005). This can seem like a paradox, for the soil function cannot be restored without a source of carbon, but a source of carbon cannot be restored without a return of soil function.

To overcome this paradox, restoration ecologists have developed several strategies. When the original top soil from a mined site cannot be replaced, the most common strategy for facilitating plant establishment on dysfunctional soils is to add amendments. For example, nitrogen and phosphorous can be added to the soil by fertilizing. However, in restored surface mines, the initial supplementation of soil nutrients for plant establishment has been found to be short lived and without a continual addition of nutrients, habitat recovery is compromised (Bradshaw 1983, Seaker and Sopper).

Ecologists may also use nurse plants to aid restorative efforts (Padilla and Pugnaire 2006). For example, shrubs may provide a benefit to newly planted tree seedlings by providing shade and minimizing evaporative water losses (Callaway 1995; Gomez-Aparaicio et al. 2004). Legumes can also facilitate long term incorporation of soil nitrogen, supporting ecosystem self-sustainability. However, the use of nurse plants such as legumes can also produce undesirable effects from competition (Blignaut and Milton 2005; Huber-Sannwald and Pyke 2005) or allelopathy (Gómez-Aparicio et al. 2004). The

most ideal strategy is to plant native colonizers which are well adapted to disturbed or oligotrophic conditions. Once established, native colonizers may provide just enough of the organic matter input to generate a nitrogen budget and jump start natural successional processes, including the proliferation of belowground biota.

As soil fertility is considered the most important limiting factor of plant growth on mined soils, nutrient availability measures should be recognized as critical in determining the progress of ecosystem recovery (Bradshaw 1983; Insam and Dosch 1988; Bradshaw 1997). Restoration ecologists have identified soil properties that serve as good indicators of ecosystem function (Bendfeldt et al. 2001). These include measures of nutrient availability (e.g. soil organic matter, total nitrogen, carbon, phosphorous etc.), microbial biomass, mycorrhizal associations, or soil structure and chemistry (aggregate stability, bulk density, water retention, conductance, and pH). Soil structure is important because it can directly influence other chemical and physical soil processes and serve as an indicator of soil degradation (Cerda 2000). Soil properties can also determine both the amount and type of soil fauna (Bass and Bischoff 2001). In addition to relating these properties with above ground structure, soil functions can be correlated with below ground structural attributes such as root density or root length.

Measurements of soil organic matter (SOM) are important because SOM serves as both an energy source for heterotrophic soil microbes and a source of mineralizable nitrogen. When microbes are not limited by nitrogen, SOM is broken down (decomposed), which releases dissolved organic nitrogen (DON) into the soil. This DON is then mineralized to ammonium (NH_4^+), which becomes available for plant uptake. The DON can also be taken up directly by microbes or plants through their mycorrhizal

associations (Jones et al. 2005). When microbes are nitrogen limited, they reabsorb DON, immobilizing nitrogen for their own metabolic requirements. Plants can also assimilate nitrogen in the form of nitrate (NO_3^-). Nitrate forms from ammonium (NH_4^+) during nitrification, but also enters the ecosystem through atmospheric deposition (Campbell et al. 2000). Stable ecosystems are thought of as closed systems since the inputs and outputs of nutrients are minimal with respect to the amount that is internally cycled (Chapin et al. 2002). Measuring nutrient availability in the soil is an indirect way of quantifying nutrient cycling and assessing ecosystem stability (Insam and Dosch 1988; Kelly and Harwell 1990; Fuhlendorf et al. 2002).

Microbial biomass is another way to monitor ecosystem recovery since it is also considered a good bio-indicator of soil function (Powlson et al., 1987; De Deyn 2003). Even though microbial biomass only comprises 1-3% of the total soil carbon and 5% of the total soil nitrogen, it plays a major role in maintaining soil fertility (Jenkinson and Ladd 1981). This is because microbial biomass is considered the most labile source of nutrients (Martens 1995; Templer 2003). Also, microbial biomass increases with increasing ecosystem complexity. For example, on retreating glacier moraines, Ohtomen et al. (1999) found increases in microbial biomass were related to plant diversity. Similarly, Zak et al. (1990) demonstrated microbial biomass was positively correlated with primary productivity in late successional ecosystems. Microbial diversity can also increase with succession (Schipper et al. 2001; Nemergut 2007). However, measuring microbial diversity (e.g. species composition) for assessing soil function may not be as good an indicator as microbial biomass given the redundancy of functional groups in soil biota (Wardle et al. 2004). In other words, dissimilarity in microbial species composition

between two sites may not necessarily suggest differences in ecosystem functioning because different microbial taxa may perform comparable functions (Harris 2003). Therefore, because microbial turnover is a main driver in nutrient availability, the amounts of microbial carbon or nitrogen are better indicators of function than measures of microbial diversity.

Mycorrhizal associations can also provide information regarding ecosystem recovery. The mutual association between plant roots and symbiotic fungi can be found in almost every type of terrestrial ecosystem (Smith and Read 1997). Plants are better able to exploit soil resources through these associations which are especially important in oligotrophic environments. These associations represent an optimization of nutrient flow between plants and soil and thus can be considered indicative of ecosystem stability (van der Heijden et al. 1996).

In addition to chemical and biological soil measures, physical measures of the soil also provide valuable information regarding soil development and ecosystem recovery. For example, structural properties such as aggregate stability are a good measure of resilience to erosive forces. Likewise, soil bulk density, a measure of compaction, will affect the rate at which roots and moisture will penetrate the soil. Compacted soils are less porous and restrict the flow of nutrients to the rhizosphere. Soil properties such as these can affect plant and root growth. Comparing structural soil attributes in a restored ecosystem with those of a reference system can give the restoration practitioner a good indication of the progress of soil development and ecosystem stability.

The allocation of nutrients in an ecosystem reflects ecosystem stability. However, nutrient dynamics are driven by many factors which vary spatially and temporally.

Nutrient dynamics are highly influenced by temperature or climate (Maithani 1996), water availability (Devi and Yadava 2006), primary productivity (Robertson et al. 1997), and floral or faunal diversity (Sedia and Ehrenfeld 2005). Consequently, samples collected during one part of the year may differ from those measured during another part of the year. Also, in some terrestrial ecosystems, nitrogen allocation may be temporally partitioned between plants and microbes (Chapin et al. 2002). In temperate environments during periods of plant dormancy, microbial activity results in pools of nitrogen that are available to the plants as they enter the growing season. During periods of plant growth, microbial biomass decreases as nutrient competition between plants and microbes increases. Not only do nutrient cycles vary temporally across ecosystems, they can also vary on a relatively small scale (Cain et al. 1999). Nutrient pools may develop under plants differently than in adjacent bare areas (Derner and Briske 1999). Although there are many factors to consider when understanding nutrient dynamics, general patterns in nitrogen mineralization or immobilization may emerge and can be interpreted in the context of the ecosystem being studied. It is these patterns that may be useful in evaluating ecosystem recovery of a restored site. Restored sites, which resemble reference or target sites in terms of above ground structural complexity, may be driven by functional similarities. Despite local resource heterogeneity, functional similarities between a restored site and its reference can serve as a useful tool in evaluating whether a recovering ecosystem responds comparably to abiotic or biotic controls and follow similar successional trajectories.

Evaluating the recovery of below ground function is especially important in oligotrophic environments such as those of the New Jersey Pine Barrens. Considered a

"leaky" ecosystem in which sandy soils cause high rates of leaching and poor nutrient retention, the conservation of nutrients in these environments is imperative for sustainability. This region also supports a globally rare forest dominated by a diminutive form of the pitch pine, *Pinus rigida*. There are many hypotheses to explain the diminutive form of the *P. rigida*, but most ecologists agree the pygmy form evolved as a result of the historically high frequency of wildfires (Boyd 2008). The pine plains, as these forests are known, have been impacted by both gravel mining and military training operations. Restoring the pine plains is a high priority for conservation efforts because this unique ecosystem is globally imperiled.

Since the late 1980's, there have been many attempts to restore mechanically disturbed areas of the pine plains. Early trials involving transplantation of nursery-grown *P. rigida* onto plowed or excavated soils at the Warren Grove Gunnery Range (WGR) showed good recovery, but only when treated with composted sewage sludge (Fimbel and Kuser 1993). In the absence of heavy organic amendments, growth and recovery of *P. rigida* was limited. Because treated sludge was later prohibited for use in the Pine Barrens, reforestation efforts in the pine plains resulted in slow growing, thin, monocultures of pitch pine (Chapter 2). As an alternative to reforestation, the planting of warm season native grasses were investigated (Chapter 3). These studies demonstrated that revegetating disturbed sites with warm season grasses facilitated natural recovery as compared with a disturbed site undergoing natural succession. While the previous studies evaluated a restored gravel pit in terms of above ground structure the purpose of this study was to identify similarities in below ground functional recovery. Based on finding similarities in plant community structure above ground, I hypothesized there would be no

differences in soil processes between the restored site and naturally recovering reference site. Specifically, I tested the hypotheses that nitrogen allocation and root growth patterns on a restored gravel pit in the pine plains would model those occurring on a naturally recovering disturbed pine plains site.

METHODS

1. Study Area:

This field study was performed at the New Jersey Air National Guard's Warren Grove Gunnery Range (WGR) in Burlington County, New Jersey, U.S.A. (lat 39°41'N, long 74°23'W) (Appendix 1). WGR is a military training site located in the East Plains portion of the Pinelands National Reserve (PNR). Occupying approximately 22% of New Jersey's total land area, the Pinelands National Reserve protects and manages roughly 450,000 hectares of pine barren habitats. The Pine Barrens are located on the outer Atlantic Coastal Plain, characterized by unconsolidated sandy soils deposited from a cyclic rising and subsiding of sea level over geological time. Approximately 13 to 16 soil types have been identified in the Pine Barrens which are characteristically acidic, well-drained and nutrient-poor (Markley 1998). The ecosystem also experiences a high frequency of fires, which shape a unique plant community of fire-adapted species (Lutz 1934; Good and Good 1975).

The New Jersey Pine Barrens are divided into two complexes: uplands and lowlands, of which there are multiple forest types (McCormick 1998; Boyd 2008). Of the upland forest types, the pine plains have gained special attention worldwide because they are recognized as being globally imperiled (Natureserve 2010). WGR has approximately

1200 hectares of pine plains, a forest dominated by a short statured form of *P. rigida*. The New Jersey Air National Guard (NJANG) considers the pine plains a high priority for conservation and preservation because of its global status.

Three sites at WGR were used in this comparative analysis; a restored gravel pit (RGP), an unrestored gravel pit (UGP), and a naturally recovering target site (TS). Specific locations of all three sites are shown in Figures 35 and 36.

The restored site (RGP) is a 0.7 hectare portion of a 2.4 hectare gravel pit abandoned sometime between 1974 and 1976 (Figure 36). Sometime soon after abandonment, the site was used by the military as a Helicopter Landing Zone (HLZ). However, by the late 1990's the highly eroded surface became unsafe for military training, and it ceased to be used for such purposes. Subsequently, restorative measures were taken to reclaim the HLZ, referred to in this study as the restored gravel pit (RGP). A detailed description of the restoration methods can be reviewed in Chapter 3. While the RGP recovered, a moratorium on military activity was placed on the site. It was extended indefinitely as the RGP began to show potential for complete ecosystem recovery. After evaluating past restoration projects at WGR in 2003 and determining reforestation methods were ineffective at restoring floral complexity (Chapter 2), the revegetation methods used on the RGP were identified as an viable alternative approach for ecosystem recovery in the pine plains.

The reference site used for this study was part of a larger air-to-ground target site withdrawn from military operation sometime around 1980. The reference site (TS) was approximately 0.8 hectares and located 1 km from the RGP (Figure 35). After being withdrawn from military operations, the disturbed TS showed signs of natural recovery.

Because the TS was undergoing succession without any intervention (spontaneous succession), it was used as a reference site for pine plains habitats naturally recovering from mechanical disturbances.

In order to characterize soil function in unproductive or dysfunctional pine plains systems, a third site was included in this comparative analysis. A 0.9 hectare unrestored portion of the abandoned gravel pit (UGP) was used as a pre-restored baseline for the RGP. In the 32 years since abandonment, the UGP received no restorative intervention and was absent of any vegetation at the time of this study (Figure 36).

2. Sampling and Data Collection:

In 2005, I installed six 5 x 10 m main plots in each of the three sites (RGP, UGP, TS) for a total of 18 main plots. Within a random subset of nine main plots (3 from each site), I randomly nested nine 1 m^2 sub-plots (three per primary plot), for a total of 27 sub-plots among the three sites. I assessed soil chemistry three times during the 2005 growing season (early July, late August, and mid October) by removing from each sub-plot two 10 cm soil cores with a 5 cm diameter soil borer for a total of 18 cores per site. Paired cores were then combined into a single sample. This resulted in a total of 27 samples for each sampling period. This protocol was chosen because 27 was the maximum number of samples that could be collected and processed in the laboratory in the same 24-hour period. Samples were bagged in the field, refrigerated, and returned to the Rutgers Field Station for processing of extractable NO_3^-, NH_4^+, dissolved organic nitrogen (DON), and microbial biomass nitrogen (MBN). Every attempt was made to remove the soil cores from bare areas within the subplots. Any vegetation, if present at the point of sampling, was scraped from the surface before soil core was removed. An additional six 10 cm soil

cores were separately collected at random from within main plots at each site to determine soil pH, particle size, bulk density, and moisture content. Root length density (cm root/cm^3 soil) was determined by collecting an additional 18 soil cores from both the RGP and TS. Root length soil cores were not collected from the UGP once it had been verified that roots were not present on the site. In order to minimize the edge effect of plants bordering the main plot, all soil sampling was conducted within a 0.5 m buffer on the interior of the plot. This provided a total sampling area of $36m^2$ ($9m^2$ x $4m^2$).

At the end of the growing season, root and shoot characteristics of *Schizachyrium. scoparium* (root and shoot mass, mycorrhizal colonization, foliar C:N) were determined by harvesting all individual clones within 10 randomly selected $1m^2$ subplots within the main plots of both the RGP and TS. The UGP was excluded because there were no plants present. All plants within the selected subplot were carefully harvested to minimize any loss of fine root mass. Plants were bagged and returned to the laboratory where roots and shoots were separated and dried. Plant community structure of the sites were collected the same year from the 5x10m main plots and analyzed in a separate study (Chapter 3).

3. Soil Properties:

Soil pH was analyzed by making a 3:1 slurry solution of deionized water with dry material and measured with a pH electrode and volt meter (Accumet Gel-Filled; Orion model 720A). Particle size was determined through sieve analysis and soil type was determined using the USDA soil classification scheme (Soil Survey Staff, 1999). Soil percolation rates were measured *in situ* and moisture content was determined gravimetrically. Soil samples were dried for a minimum of 48 hours at 70°C.

4. Plant Attributes:

Root and shoot mass were determined by weighing the dried parts from harvested *S. scoparium* plants. A subsample of fresh roots was selected at random and analyzed for mycorrhizal colonization, and a subsample of shoots were collected for foliar carbon and nitrogen content (C:N) analysis. Foliar carbon and nitrogen content were assayed by high temperature combustion (Leco TruSpec carbon/nitrogen determinator). Mycorrhizal colonization was assessed following a method described by Koske and Gemma (1989). Roots were first rinsed and fixed in 50% ethanol before clearing with a 10% KOH solution. Samples were then acidified with 1% HCl before being stained with 0.05% trypan blue for 20 minutes over low heat. I then examined the stained roots under a compound microscope (40x), and quantified colonization using a modified gridline intercept method (Giovanetti and Mosse 1980).

I measured root length through image analysis. Root samples, obtained from soil cores, were serial wet sieved in 5 fractions, and air dried for 24 hours. Roots were then scanned and measured using Image-Pro Express 4.1 Software (Media Cybernetics, 1999). Root length density was determined by dividing root length by soil sample volume.

5. Soil Chemistry:

a. Inorganic Nitrogen

Soil bound inorganic NH_4^+ was measured using an ion selective electrode (Orion model 95-12 ammonia electrode, Orion Research Inc Boston MA) and NO_3^- was measured by ion chromatography (Dionex, DX100). Extraction methods for ammonium

followed those of Foster (1995). Fresh soil samples (15g) underwent extraction with 0.5M solution of K_2SO_4 (30mL) before being filtered and analyzed. Similarly, a second soil sample was extracted with distilled H_2O for nitrate analysis. The NO_3^- analysis followed procedures according to Standard Method protocols (American Public Health Association 1998). All mineral nitrogen extractions occurred within 24 hours of collection.

b. Organic Nitrogen

Microbial biomass nitrogen (MBN) was determined and calculated using the chloroform fumigation-extraction method (Brooks et al. 1985). Soil samples were weighed and divided into two fractions. One fraction was subjected to chloroform fumigation for 24 hours. This fumigation killed and lysed microbial cells releasing microbial nitrogen into the soil. Samples were fumigated within four hours of collection. Both fractions were extracted with K_2SO_4 and then subjected to an acidic Kjeldahl digestion where, in the presence of a catalyst (selenium), acid (H_2SO_4) converted organic nitrogen (microbial and dissolved) into ammonium. The digested sample was then analyzed with an ion selective electrode for ammonium. The fumigated fraction contained soil-bound inorganic NH_4^+, DON, and MBN, while the second (unfumigated) fraction only contained inorganic NH_4^+ and DON. Therefore, MBN was calculated from the difference of the two fractions multiplied by a correction factor derived from standard conditions ($k_n = 0.54$; Brooks et al. 1985). Dissolved organic nitrogen (DON) was estimated from the difference between the measured unfumigated fraction and the soil extracted NH_4^+.

6. Statistical Analysis:

Differences in soil chemistry among study group means were analyzed using a factorial analysis of variance with season and site as fixed main effects. Univariate analyses (one-way ANOVA) were performed on single event sampling soil parameters (e.g., plant attributes). The control site was excluded from analyses where data was unavailable. Data were square root transformed if they violated the assumptions of normality or homocedascity of variance. Normality of data was verified using the Shapiro-Wilk statistic, which is a robust test of normality when sample sizes are less than 200 (Shapiro and Wilk 1965). I checked homogeneity of variance using the Levine Statistic. In some cases where unequal variances could not be corrected with transformations, I used the Brown-Forsythe modified F-statistic to substitute for the ANOVA F statistic. The Brown Forsythe ANOVA can be more robust to unequal variances (Mendes 2004). *Post hoc* differences between site means were determined using Fisher's Least Significant Difference (LSD) test. In cases where variance was unequal, I replaced the LSD test with the Games-Howell *post hoc* test because it is less sensitive to heterocedascity of variances (Jaccard et al. 1984; Day and Quinn 1989).

In order to identify functional patterns among sites, I used descriptive statistics and non-parametric Spearman Rank correlation coefficients between plant and soil characteristics. Spearman ranks were chosen over parametric correlations because they do not rely on assumptions of normality and are more sensitive for identifying non-linear trends (Choi 1977). All analyses were performed using the SPSS statistical software package (SPSS Version 16.1)

RESULTS

1. Soil Properties

With the exception of soil pH, site characteristics were similar between study groups (Table 21). Soil pH was significantly higher on the RGP ($F_{[2,5.1]} = 16.6$, $p < 0.01$). All sites were primarily composed of sands (84.5% to 88.3%) but the TS had a higher silt fraction ($F_{[2,7]} = 23.9$, $p < 0.005$). Fines were negligible at all the sites.

2. Plant Attributes

Although there were no differences in mass between the roots and shoots of the RGP and TS (Figure 37), the ratio of root mass to shoot mass was significantly higher in grasses on the TS ($F_{[1,31]} = 4.338$, $p < 0.05$; Figure 38). The *S. scoparium* plants on the TS had almost 30% of their total mass allocated to root structures, while grasses on the RGP only had approximately 16% of their total weight in root mass.

There was more than 90 times total carbon than total nitrogen in above ground foliage of *S. scoparium*. However, there was no significant difference between the two sites for the ratio of total foliar carbon to nitrogen. The C:N ratios were 137.4 and 91.6 for the RGP and TS (Figure 39).

Root length per unit of soil volume was almost four times greater on the TS than the RGP ($F_{[1,16]} = 88.6$, $p < 0.001$; Figure 40). Mean root length density on the TS was 7.4 cm/cm^3 and 1.9 cm/cm^3 on the RGP. No roots were observed in soil samples of the UGP and the site was excluded from this analysis.

There were no detectable differences in the percent of grass root colonized by vesicular arbuscular-mycorrhizae (Figure 41). For both the RGP and TS, roots of individual *S. scoparium* plants had approximately 30% of their roots colonized with VA mycorrhizae.

3. Soil Chemistry

Soil nitrate (NO_3^-) varied both by site and season (Figure 42). There were significant differences among sites ($F_{[2,8]} = 17.8$, $p < 0.001$) as well as among sampling times ($F_{[2,8]} = 9.135$, $p < 0.001$). Furthermore, the factorial ANOVA showed an interaction effect for season and site ($F_{[4,8]} = 13.9$, $p < 0.001$). Soil NO_3^- on the UGP (0.93 µg/g soil) was significantly higher in July compared to both the RGP and TS which did not differ from each other (0.64 and 0.63 µg/g soil respectively) (Table 22). Soil nitrate on the UGP spiked during the month of August before decreasing in October. The RGP showed no change in levels of soil NO_3^-during the entire study, while the levels on the TS significantly decreased over the first two sampling periods before leveling off by October. By the end of the growing season, only the TS had significantly lower concentrations of soil nitrate (0.36 µg/g soil vs. 0.65 µg/g soil [UGP] and 0.63 µg/g soil [RGP]). All three sites displayed different patterns of soil nitrate over time.

The results of the factorial ANOVA comparing soil ammonium (NH_4^+) availability among sites over the growing period was also significant (Figure 43; Table 23). There was only a main effect of season ($F_{[2,8]} = 4.879$, $p < 0.05$) no effect by site, and no interacting effects of site and season. Differences in soil NH_4^+ among sites were only observed in October when the UGP (0.32 µg/g soil) and TS (0.25 µg/g soil) were

significantly lower than the RGP (0.47µg/g soil). Although all sites showed increases in NH_4^+ from July to August, only the UGP was significantly higher. The RGP showed no significant changes in NH_4^+ during the entire growing season while soil NH_4^+ on the TS significantly decreased between August (0.42 µg/g) and October (0.25 µg/g soil).

Microbial biomass nitrogen differed among sites ($F_{[2,8]} = 17.8$, $p < 0.001$) but not by season (Figure 44). The lowest levels of MBN were measured on the UGP at every sampling date (Table 24). Similarly, there were minimal changes in MBN over time on the RGP and TS. Mean MBN levels for the RGP were 9.9 µg/g soil in July, 5.2 µg/g soil in August, and 7.1 µg/g soil in October. Mean MBN levels for the TS were 8.1µg/g soil in July, 3.5µg/g soil in August, and 10.2 µg/g soil in October. Only the TS had a significant decrease in the amount of MBN during August. Despite the decrease in MBN observed on the RGP in August, the amount did not significantly differ from the other sampling times.

There were significant differences in dissolved organic nitrogen (DON) for both main effects. DON varied by both site ($F_{[2,8]} = 34.9$, $p < 0.001$) and by season ($F_{[2,8]} = 4.1$, $p < 0.05$; Figure 45). Dissolved organic nitrogen differed between sites for each month except in October; at which time only the UGP was significantly lower (Table 25). DON levels on the TS site during the month of July were twice that of those observed on the UGP (5.4 µg / g soil vs. 2.6 µg / g soil). Even though there were differences between sites at each sampling time, amounts of DON specific to each site did not change over the course of the study.

Spearman rank correlations for plant attributes are shown in Table 26. Correlations between plant level measures varied between RGP and TS. On the RGP site

both root and shoot mass were strongly positively correlated with the ratio of carbon to nitrogen in leaf tissue. The same was not true in *S. scoparium* plants of the TS. On the TS, root and shoot mass showed a significantly positive correlation with the intensity of mycorrhizal association. Both sites also showed very strong correlations between root mass and shoot to root ratios. On the RGP, the association between shoot mass and root to shoot ratios was also significant, albeit weak. There was no such correlation on the TS.

Spearman rank correlations of soil nutrient levels for each site delineated by season are shown in Table 27. There were few associations between the concentrations of nitrogen containing organic and inorganic nutrients. Only nutrients in the TS had clear associations at all three sampling periods. In July, low levels of soil NO_3^- in the TS were associated with high levels of both MBN (rho = -0.900, $p < 0.05$) and DON (rho = -0.999, $p < 0.01$). No similar associations were seen in either the RGP or UGP during the same time period. There was a negative correlation between soil NH_4^+ and MBN on the TS in August (rho = -0.762, $p < 0.05$). Only during the last sampling event were associations observed on the RGP. However these associations differed from those observed in the TS. In October, soil ammonium levels on the RGP were positively correlated with MBN (rho = 0.811, $p < 0.05$) and soil DON (rho = 0.742, $p < 0.05$) concentrations. At the TS, also during October, high levels of DON were associated with high levels of MBN (rho = 0.738., $p < 0.05$) and NO_3^- (rho = 0.700, $p < 0.05$). There were no associative patterns of soil nitrogen observed on the UGP at any time.

DISCUSSION

In Chapter 3 I showed that the plant community structure between the RGP and TS were similar in terms of vegetative cover and composition. This study also showed that density and standing biomass of the seeded native warm season grass, *S. scoparium*, on the RGP were comparable to those observed on the TS. Furthermore, considering differences in recovery times, the natural recruitment pattern of pine trees on both the restored gravel pit and naturally recovering target site appeared to follow similar successional trajectories. Given the similarities in above ground structure between the two sites, and the interdependency of ecosystem structure and function, I hypothesized that below ground function would be similar as well. Therefore, I used both nutrient availability and root and shoot growth as surrogates for ecosystem function to compare recovery between the two sites. I found some similarities between the two sites, but determining whether the sites are following similar functional recovery pathways was not as apparent.

Of the eight soil characteristics measured at each site, only two measures were different among sites. First, soil pH on the RGP was significantly higher than that of the UGP or TS. The decreased soil acidity may have explained the higher percentage of non-native plants on the RGP compared to the TS (Kourtev et al. 1998). Given that the soil pH on the adjacent UGP was more acidic and similar to the TS, the increase in pH on RGP likely occurred during soil preparation. It is not entirely clear why restoration caused an increase in soil pH since others have found that the addition of organic matter reduced soil pH (Yan 1996; Chapin et al 2002). An increase in soil pH on the RGP however, may have been related to the use of fertilizers (Ballard 2000) or indigenous

backfill material which, upon weathering, may have buffered the soil. It's also possible for invasive plants to alter soil pH (Kourtev et al. 2003). Because soil pH affects nutrient cycling, nutrient availability, plant productivity and even mycorrhizal function, differences in soil acidity must be considered when comparing soil function.

Second, the soil clay fraction was higher on the TS than the RGP and UGP. Because particle size and structure can influence biochemical and geochemical processes, differences in soil structure has the potential to cause differences in ecosystem structure and function (Ladd et al. 1996). It is plausible that the spontaneous recovery observed on the reference site was enhanced in part by the increase in the clay fraction measured in soils of the target site. Clay can not only improve nutrient availability through increased cation exchange, but can also increase soil moisture retention (Sawney 1972; Hudson 1994). Additionally, because the TS was only surface plowed and not strip mined, it is likely a seed bank also remained intact. Given all these conditions, it is perhaps not surprising that spontaneous succession occurred on the TS. Conversely, with a dysfunctional soil profile and a missing seed bank on the abandoned gravel pit, spontaneous succession was inhibited, requiring a restorative intervention to facilitate recovery. While the differences in soil properties between the TS and UGP were minimal, The UGP represents a dysfunctional pine plains system and the differences cannot be underestimated in understanding the limitations to spontaneous recovery in this ecosystem.

Although there were no differences in root or shoot biomass between the RGP and TS, the ratio of root mass to shoot mass on the RGP was less than that of the TS. Differences in shoot to mass ratios were likely due to the greater range of plant sizes

observed on the RGP. Shoot mass ranged from 0.05g to 117.3g on the RGP compared to 0.02g to 38.26g on the TS. This clearly indicates greater variability in above ground size. When calculated as a ratio with root mass, the greater variability produced differences in root to shoot ratios between sites, which were not detected by shoot mass alone. The root to shoot mass data suggests that plants on the RGP required less root mass to support aboveground plant growth when compared with plants on the TS. Alternatively, root growth for restored grasses required greater shoot productivity. Given that roots provide nitrogen and other important nutrients for above ground structures, and shoots provide the necessary carbon building blocks for below ground structures, the balance of root to shoot mass is essentially a balance of plant resources between the two structures (Davidson 1969; Agren and Ingestad 1987).

The optimal resource use theory predicts that plant growth will be controlled by finding the equilibrium between limited light and nutrient availability (Bloom et al. 1985; Tilman 1988). In order to maintain this balance, growth will be greater in the structures that acquire the limiting resource (Aikio and Markkola 2002). Thus, root to shoot data might indicate grasses at the RGP were light limited when compared with grasses of the TS. In fact, grasses on the RGP were much taller than TS grasses (see Chapter 3, Figure 11) which could have produced shading effects allowing even greater above ground growth (Wilson 1988). The greater shoot mass observed on the RGP however may not be indicative of increased nutrient availability as soil nitrogen levels, discussed below, do not necessarily support this. Therefore, it is not entirely clear whether grasses on the RGP allocated more energy to above ground growth because they were light limited. Alternatively, the difference between root to shoot ratio may explain the greater

variability in shoot mass observed on the RGP. The greater variability in plant sizes on the RGP may represent functional difference between the two sites. One possible explanation for the difference is an increase in resource heterogeneity. For example, woodchips used during restoration were not uniformly distributed across the RGP and could have produced spatial differences in soil properties. In a controlled germination study of *S. scoparium*, differences in wood chip depth resulted in different growth responses (Chapter 4). The greater variability in plant growth observed on the RGP may have been a result of the sampling method, which did not take into consideration the variations in wood chip cover across the site. Had woodchips been more evenly distributed across the RGP during restoration, a root to shoot difference may not have been measured. However, it has been shown that environmental heterogeneity influences ecosystem structure and function (Tilman 1982; Ettema and Wardle 2002; Wijesinghe et al 2004; Hutchings et al. 2003) and this patchiness may have actually been beneficial for supporting natural recruitment of other plants.

Root abundance is important for water and nutrient uptake. Differences in root density between the RGP and TS may suggest differences in soil function, differences in resource availability, or differences in aboveground structural complexity. Several factors can influence root length density. For example, soil bulk density will limit root penetration thus decreasing root lengths (Zimmerman and Kardos 1961; Heilman 1981; Unger and Kaspar 1994). Nutrient availability can also affect root growth since roots may proliferate in nutrient rich patches (Crick and Grime 1987). Whether increases in root length density result in greater aboveground biomass is less clear. For this study, pines on the RGP were larger than pines on the TS largely due to differences in recovery times.

The increase in root density may be related to above ground biomass. However, Sainju and Good (1993) found that in the pine plains, root density was not correlated with tree size. They also found that root density was positively correlated with total soil nitrogen which may suggest that in oligotrophic systems like the pine plains, plants may develop more extensive root systems to maximize nutrient uptake. Ehrenfeld et al. (1992) observed decreased root mass in the pine plains, but the highest relative quantities of roots were observed in mineral soil of four studied pine barren forests. Increased root density may increase resource competition between plants and alter nutrient availability in the rhizosphere. Parmelee et al. (1993) reported that extractable soil nitrogen decreased with increasing root density of *P. rigida*. Although differences in root density between sites may suggest differences in nutrient availability, they can also be indicative of species specific responses to resource heterogeneity (Fransen et al. 1998; Crick and Grime 1987).

Litter quality, typically described by the ratio of carbon to nitrogen concentration in leaf matter (C:N), plays an important role in moderating decomposition and mineralization rates (Chapin et al. 1988). When C:N ratios are low, microbes readily decompose the organic matter because there is enough nitrogen for microbial processes. However, at high C:N ratios, microbes become nitrogen limited and begin removing inorganic nitrogen from the soil (immobilization). The threshold for net nitrogen mineralization occurs when litter has a C:N ratio of 25:1 or less (Chapin et al. 1988). In this study, the C:N ratio of *S. scoparium* tiller tissue was greater than 25:1 in both sites, thus it is plausible that *S. scoparium* litter alone did not provide the nitrogen requirement for microbial breakdown of organic matter, and soil microbes were competing with plants

for soil nitrogen. *Schizachyrium scoparium* does well in nitrogen limited environments because it has high nitrogen use efficiency (NUE) (Wedin and Tilman 1990). Along an increasing nitrogen gradient, foliar C:N in *S. scoparium* declines and *S. scoparium* becomes outcompeted by other grasses (Wedin and Tilman 1996). The similarity in foliar C:N ratio between sites suggests similarities in nutrient availability.

There were no differences between the RGP and TS in the amount of root length colonized with mycorrhizae. Plants benefit from a symbiotic relationship with mycorrhizae through enhanced absorption of organic and inorganic nutrients. Mycorrhizal associations are found in almost every type of terrestrial ecosystem, but type and extent of association can vary (Slankis 1974; Smith and Read 1997). The percentage of root lengths colonized by mycorrhizae is dynamic and fluctuates with changing environmental conditions (Slankis 1974). Colonization rates can vary due to moisture, nutrient availability, leaf litter quality, disturbance or successional age (Anderson and Liberta 1992; Johnson et al. 1992; Eom et al. 1999; Conn and Dighton 2000; Blanke et al. 2005). Although this association enhances uptake of multiple limiting nutrients, these associations may be most important in phosphorous uptake (Gange et al 1999; Allen 2003; Hodge et al. 2000). Similarities in mycorrhizal colonization rates may indicate comparable responses to environmental changes. For example, it has been shown that colonization rates will decrease over an increasing nitrogen gradient (Treseder and Allen 2002; Dighton et al. 2004). *Schizachyrium scoparium* forms obligate associations with vesicular-arbuscular mycorrhizae (VAM) which penetrate and grow in cell walls of the root cortex. The mycorrhizal colonization rates observed in S.*scoparium* on the restored and reference site are consistent with what has been reported in other studies (Wallace

1987; Anderson and Liberta 1992). Given that rates were measured once in this study, it remains unclear whether mycorrhizal dynamics between the RGP and TS site are similar. However, based on the similarity of root length infected with mycorrhizae at the end of the growing season, the nutrient status of planted and naturally recruited *S. scoparium* appeared comparable.

It is well known that nutrient dynamics vary temporally and spatially and are influenced by species composition (Vitousek et al. 1982; Robertson et al. 1988; Robertson et al. 1997; Hobbie 1992; Sedia and Ehrenfeld 2005; Eviner et al. 2006). Furthermore, many ecosystems show a partitioning of nitrogen assimilation between plants and microbes during the year (Jaeger et al. 1999). Similar trends were observed on all three sites in this study. The flux of organic and inorganic nitrogen varied among sites and across the growing season. A seasonal and site effect was measured for soil NO_3^-. Temporal changes were seen on both the UGP and TS, while soil NO_3^- on the RGP remained relatively constant throughout the study. Soil NO_3^- on the TS was highest in July and decreased through summer and into fall. The NO_3^- signal on the TS is typical of many temperate forest ecosystems in which inorganic nitrogen builds up during periods of plant dormancy, but is then depleted during the growing season (Jaeger et al. 1999). However, in grasslands NO_3^- uptake is typically greatest in spring (Jackson et al. 1988; Zhang et al. 2008). Neither pattern however was observed on the RGP as soil NO_3^- remained constant throughout the growing season. These data obscure whether nitrogen allocation on the RGP resembles a forest or grassland system, or some intermediate successional stage. Given that the RGP was amended with woodchips, I would have expected an increase in microbial nitrogen demand to decompose the added carbon.

Torok et al. (2000) found a similar nitrogen demand pattern in restored grasslands where carbon amendments were used. However, the constant NO_3^- supply measured on the RGP throughout the season has been observed in other forests types (Nadelhoffer et al. 1984). These data suggest that, during the growing season, mineralization plays more of a role than nitrification, especially since the formation of ammonium precedes nitrate and there is a high energetic cost to reduce NO_3^- in plants (Gutshik 1981). The pulse of NO_3^- observed on the UGP during August and resolved by October has been observed in other studies (Lipson et al. 1999). It may be related to the decrease in MBN during peak growth periods. There were no plants on the UGP to assimilate NO_3^-, thus it remained high relative to the vegetated sites. Gray and Dighton (2009) observed similar trends in nitrification on soil samples from treeless plots. By the end of the growing season, soil nitrate on the control site decreased. Nitrate was either lost through leaching or assimilated by the microbial community as the N requirement increased with increasing availability of organic matter from microbial turnover. Extractable soil nitrogen is susceptible to leaching in Pine Barren systems, especially in early successional sites (Woodwell 1998).

There were no differences in soil ammonium (NH_4^+) among sites in July and August. In October, NH_4^+ on the TS was significantly lower than NH_4^+ on the RGP and UGP. Soil NH_4^+ increased between July and August and decreased in October on the TS and UGP but not the RGP. Soil NH_4^+ reached its lowest levels on the TS in October, while soil NH_4^+ reached its highest level on the UGP in August. When microbes are not limited by nitrogen, organic nitrogen is mineralized. If more NH_4^+ is being produced than what is assimilated, net mineralization occurs. Because soil NH_4^+ on the RGP and

TS did not change between July and August, NH_4^+ was being scavenged by plants and microbes as quickly as it was being produced. In contrast, from August to October, NH_4^+ on the TS decreased suggesting nitrogen was being immobilized from the soil. Alternatively, NH_4^+ was converted to NO_3^- by nitrifying bacteria thus the fate of NH_4^+ on the TS cannot be ascertained with certainty. However, the drop in NH_4^+ from August to October without a corresponding increase in NO_3^- during that time suggests nitrification was minimal compared to immobilization. The decrease in NH_4^+ during October was less evident on the RGP and may be a result of the plant community being more developed on the TS than the RGP, where nutrient cycling may have been more efficient (Tilman et al. 1996; Loreau et al. 2001).

The increase in NH_4^+ on the UGP in August suggests that net mineralization had occurred on this site but these results seem counterintuitive since there was less organic matter to decompose. With less organic matter and lower microbial biomass on the UGP, I would have expected decomposition to be limited and NH_4^+ pools to be lower than those of the vegetated sites (TS or RGP). If however microbial turnover is high, as a result of decreased water availability on the bare substrate, elevated NH_4^+ pools are possible as microbes compete heavily for carbon (Kaye and Hart 1997; Schmidt et al. 2007).

The levels of MBN on the RGP and TS had a bimodal pattern and were higher in the early summer, decreased by late summer, and then increased during the fall. The MBN was significantly lower in the UGP for the same time period. The decrease in MBN at the height of the growing season has been observed in other terrestrial systems and is thought a consequence of competition from increased plant growth after a period of

dormancy (Garcia and Rice 1994; Lipson et al. 1999; Jaeger et al. 1999) or related to abiotic factors (Maithani et al. 1996; Bass and Bischoff 2001; Devi and Yadava 2006). The MBN is a sink for nitrogen when plant growth is minimal. Mineralization occurs during peak growing periods and during microbial turnover (Singh et al. 1989; Zak et al. 1990). This may explain why soil NH_4^+ on the TS was negatively correlated with MBN. High levels of MBN at other times during the year may have indicated the ability of ecosystems to tightly store nitrogen that would otherwise be lost (Holmes and Zak 1999).

Given the presence of soil MBN in the UGP, and low levels of inorganic nutrients throughout the growing season, it would be incorrect to refer to this site as biologically inactive. However, there were no clear relationships between inorganic and organic nitrogen forms on the UGP. The potential for plant growth may not have been limited by nutrients alone. Xeric conditions may have been the single most constraining factor inhibiting spontaneous succession in highly disturbed pine plains sites as it was observed in other dysfunctional sites (Bishop and Chapin 1989; Wood and Del Moral 1997; Walker and Del Moral 2003; Moore et al. 2006).

As microbes break down soil organic matter, dissolved organic nitrogen (DON) is released. DON can also enter the system through precipitation, as water comes in contact with organic matter. Dissolved organic nitrogen can bypass mineralization through direct uptake by microbes and plants (Chapin et al. 2002). The amount of DON in the soil can then serve as an indicator of the amount of soil organic matter. There is growing evidence that the greatest losses of terrestrial nitrogen are from leaching DON (Neff et al. 2003). In this study, the amount of DON in the soil remained unchanged at each site over the course of the study which is consistent with other forest systems (Campbell et al. 2000).

Dissolved organic nitrogen fluxes may be more tightly coupled with abiotic factors (e.g. precipitation) than any other factor (Neff et al. 2003). All sites had seasonal differences in DON with the exception of the TS and RGP which were not statistically different in October. The UGP had the lowest DON while the TS had the greatest DON. Since the main source of soil organic matter is from decomposing roots (Coleman et al. 2004), DON was greatest on the TS because it had the greatest root density. It is not clear why a significant increase in DON was observed on the RGP in October but may be related to the added soil amendments. This increase in DON at the end of the growing season has been seen in other terrestrial systems and may be related to increases in litter and decomposition (Neff et al. 2003). Although the amount of DON in the soil can be influenced by litter quality, this may not have been the case given foliar C:N ratios of the dominant grass did not differ between the TS and RGP. However, without measuring litter quality directly this cannot be determined.

In October, DON on the TS was positively correlated with levels of NO_3^- and MBN, but DON on the RGP was positively correlated with NH_4^+. These correlations suggest that the fate of DON was different between the two sites. However, the specific reasons for these differences remain unclear as there are many factors that influence cycling of organic nitrogen (Chapin et al. 2002; Coleman et al. 2004).

Because I could only test for differences between sites and not necessarily differences in experimental treatment, it was difficult to avoid the problem of pseudoreplication (Hurlbert 1984; Oksanen 2001). Although experimental treatments were not interspersed in space, I assumed both sites were capable of sustaining pine plains forests because they were geologically similar, located within 1 km of each other,

and surrounded by similarly intact pine plains communities. However, each site had different disturbance histories and it is unclear if the same restorative treatment would be as effective if applied to other disturbed sites. Additionally, this study did not address the role of non-vascular plants, which can be important in influencing nutrient cycles in pine barren soils (Sedia and Ehrenfeld 2005)

Although a naturally recovering gravel pit in the pine plains would have served as an ideal reference for comparing functional recovery, I was unaware of any such site. Areas in the pine plains where soils have been excavated remain dysfunctional and absent of vegetation and require restoration. This study demonstrated in abandoned gravel pits, seeding with native grasses shows potential to repair dysfunctional nutrient cycles, improve soil development, and facilitate natural successional processes.

CONCLUSION

In this study, I compared the functional recovery of a restored gravel pit with that of a disturbed area showing signs of natural recovery. Prior studies indicated that both sites are following similar successional trajectories in terms of above ground structure. Because of the similarities in floral structure between the two sites and because of the important feedbacks driving above ground structure and below ground function, I hypothesized that soil function between the two sites would show similar patterns of recovery. I used measures of nitrogen availability in the soil as a surrogate for below ground function. I also measured root growth and nutrient status of the dominant native grass *S. scoparium*. Soil chemistry was compared with an unrestored gravel pit to provide information on pre-restored conditions. I found that plant attributes between the restored

and reference site were similar in terms of mycorrhizal colonization rates, root mass, and foliar C:N ratios. There were differences in root to shoot ratios, and root length density. Nitrogen allocation in the restored and reference site followed similar seasonal patterns, but amounts varied between sites. However, because of the spatial and temporal variability, differences between soil function may not necessarily indicate differences in ecosystem recovery, but rather the differences in recovery time between sites and the natural variability in plant-microbe interactions.

This study demonstrated that assisted succession may be the best approach to restoring highly disturbed sites in the pine plains of the New Jersey Pinelands. The establishment of warm season grasses facilitated the recovery of highly disturbed sites by improving soil conditions, restoring nutrient cycling, providing a vegetative cover to retain moisture, and creating safe sites for natural recruitment. As the RGP and TS continue to recover, these data suggest that the successional trajectories of the two sites will likely converge towards a mature pine plains habitat.

Table 21. Soil characteristics for each study site. Values are group means +/-1 S.E.M. Means with different lowercase letters indicate significant differences at $p < 0.05$.

	Restored Gravel Pit	Target Site	Unrestored Gravel Pit
pH	6.17 +/- 0.4 [b]	4.98 +/- 0.0 [a]	4.62 +/- 0.3 [a]
Bulk Density (g/cm^3)	0.71 +/- 0.2	0.67 +/- 0.2	0.67 +/- 0.1
Perc Rate (L/min)	0.33 +/- 0.6	0.25 +/- 0.4	0.25 +/- 0.1
% Moisture	4.92 +/- 0.2	4.92 +/- 0.5	5.84 +/- 0.4
Grain Size			
Gravel	15.1% +/- 1.2	12.3% +/- 2.6	11.4% +/- 2.9
Sand	84.5% +/- 1.2	85.5% +/- 2.4	88.3% +/- 2.9
Silt	0.3% +/- 0.1 [a]	1.6% +/- 0.3 [b]	0.2% +/- 0.1 [a]
Fines	0.1% +/- 0.0	0.4% +/- 0.1	0.1% +/- 0.0

Table 22. Results of the means separation post hoc examination (Games Howell) for soil NO_3^- levels between seasons (A) and between sites (B). (UGP = Unrestored Gravel Pit; RGP = Restored Gravel Pit; TS = Target Site).

A

		July	August
UGP	August	**0.009**	
	October	NS	**0.002**
RGP	August	NS	
	October	NS	NS
TS	August	**0.001**	
	October	**0.001**	NS

B

		UGP	RGP
July	RGP	**0.018**	
	TS	**0.027**	NS
August	RGP	**0.002**	
	TS	**0.001**	0.031
October	RGP	NS	
	TS	**0.034**	**0.014**

Table 23. Results of the means separation post hoc examination (Games Howell) for soil NH_4^+ between seasons (A) and between sites (B). (UGP = Unrestored Gravel Pit; RGP = Restored Gravel Pit; TS = Target Site).

A

		July	August
UGP	August	0.003	
	October	NS	NS
RGP	August	NS	
	October	NS	NS
TS	August	NS	
	October	NS	0.001

B

		UGP	RGP
July	RGP	NS	
	TS	NS	NS
August	RGP	NS	
	TS	NS	NS
October	RGP	**0.004**	
	TS	NS	**0.013**

Table 24. Results of the means separation post hoc examination (Games Howell) for soil MBN levels between seasons (A) and between sites (B). (UGP = Unrestored Gravel Pit; RGP = Restored Gravel Pit; TS = Target Site).

A

		July	August
UGP	August	NS	
	October	NS	NS
RGP	August	NS	
	October	NS	NS
TS	August	**0.032**	
	October	NS	**0.003**

B

		UGP	RGP
July	RGP	**0.001**	
	TS	**0.001**	NS
August	RGP	NS	
	TS	NS	NS
October	RGP	**0.001**	
	TS	**0.001**	NS

Table 25. Results of the means separation post hoc examination (Games Howell) for soil DON levels between seasons (A) and between sites (B). (UGP = Unrestored Gravel Pit; RGP = Restored Gravel Pit; TS = Target Site).

A

		July	August
UGP	August	NS	
	October	NS	NS
RGP	August	NS	
	October	NS	NS
TS	August	NS	
	October	NS	NS

B

		UGP	RGP
July	RGP	**0.012**	
	TS	**0.001**	**0.001**
August	RGP	**0.005**	
	TS	**0.006**	**0.005**
October	RGP	**0.029**	
	TS	**0.002**	NS

Table 26. Spearman rank correlations between plant attributes for the restored gravel pit and naturally recovering target site. * denotes significance at $p < 0.05$. ** denotes significance at $p < 0.001$.

	Foliar C:N	Mycorrhizal Colonization	Root:Shoot	Root Mass
Restored Gravel Pit				
Foliar C:N	1			
Mycorrhizal Colonization	-0.527			
Root:Shoot	-0.135	-0.177		
Root Mass	**0.879****	-0.455	-0.397	
Shoot Mass	**0.791****	-0.371	**-0.581***	**0.956****
Target Site				
Foliar C:N				
Mycorrhizal Colonization	-0.41			
Root:Shoot	-0.336	0.129		
Root Mass	0.462	**0.583***	-0.118	
Shoot Mass	0.473	**0.486***	-0.248	**0.975****

Table 27. Spearman rank correlations of soil nutrients for the restored gravel pit, naturally recovering target site and the unrestored gravel pit. ***** denotes significance at $p < 0.05$. ****** denotes significance at $p < 0.001$.

	July			*August*			*October*		
	NO_3-	NH_4+	MBN	NO_3-	NH_4+	MBN	NO_3-	NH_4+	MBN
Restored Gravel Pit									
NH_4+	-0.417			0.083			0.114		
MBN	0.083	0.400		0.000	-0.214		0.273	**0.811***	
DON	0.483	-0.033	-0.100	0.117	-0.067	0.286	-0.480	**0.742***	0.748
Target Site									
NH_4+	0.086			-0.119			-0.333		
MBN	**-0.900***	0.190		0.429	**-0.762***		0.381	-0.238	
DON	**-0.999****	-0.267	0.476	-0.571	0.400	-0.690	**0.700***	-0.333	**0.738***
Unrestored Gravel Pit									
NH_4+	0.333			-0.048			0.300		
MBN	-0.145	-0.234		0.162	-0.522		0.69	0.595	
DON	0.071	0.500	-0.577	-0.05	-0.262	-0.162	0.024	-0.69	-0.429

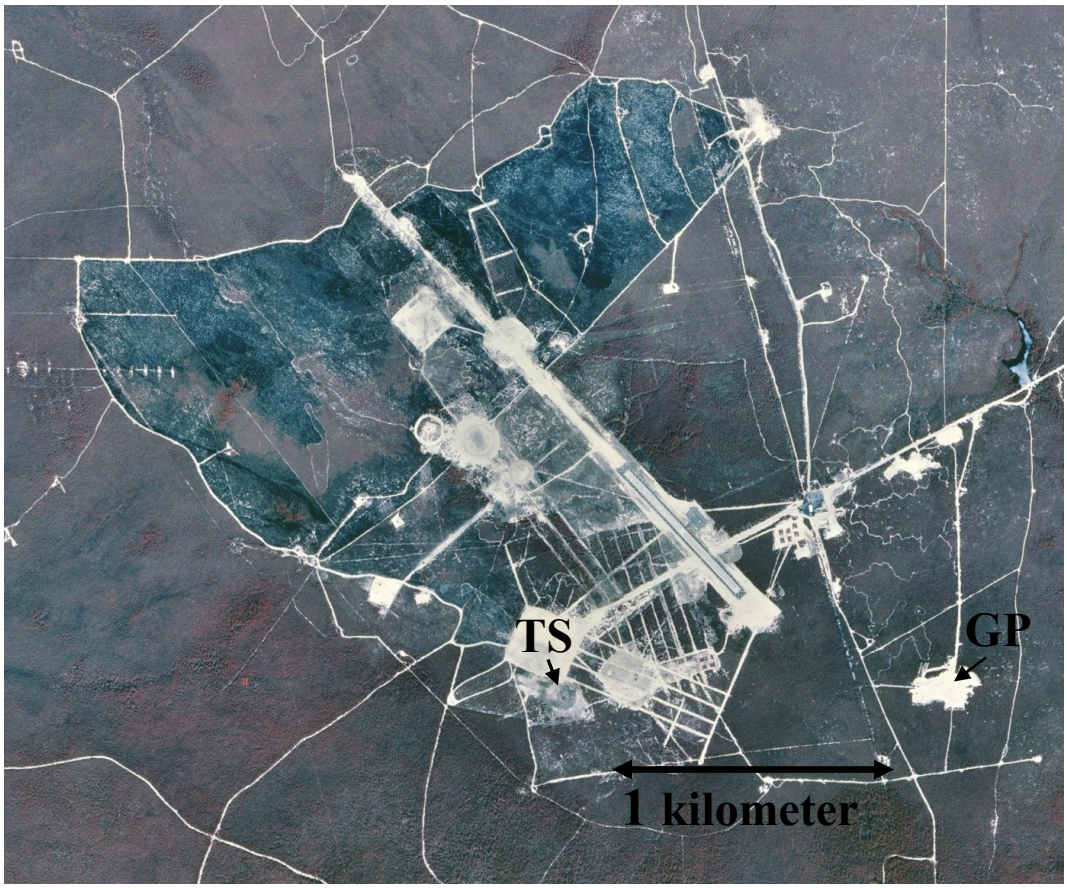

Figure 35. The target area at WGR with location of the restored and unrestored gravel pit (GP) and the naturally recovering target site (TS).

Figure 36. Aerial view of the abandoned gravel pit. Area 1 is the helicopter landing zone (HLZ) restored in 2001 by seeding with warm season grasses (*S. scoparium*) and incorporation of woodchips. Area 2 is the unrestored portion of the gravel pit used as a control for this study. Area 3 is the portion of the abandoned gravel pit restored in 1997 with pine tree seedlings. Area 4 was the site of the *S. scoparium* germination study (Chapter 4).

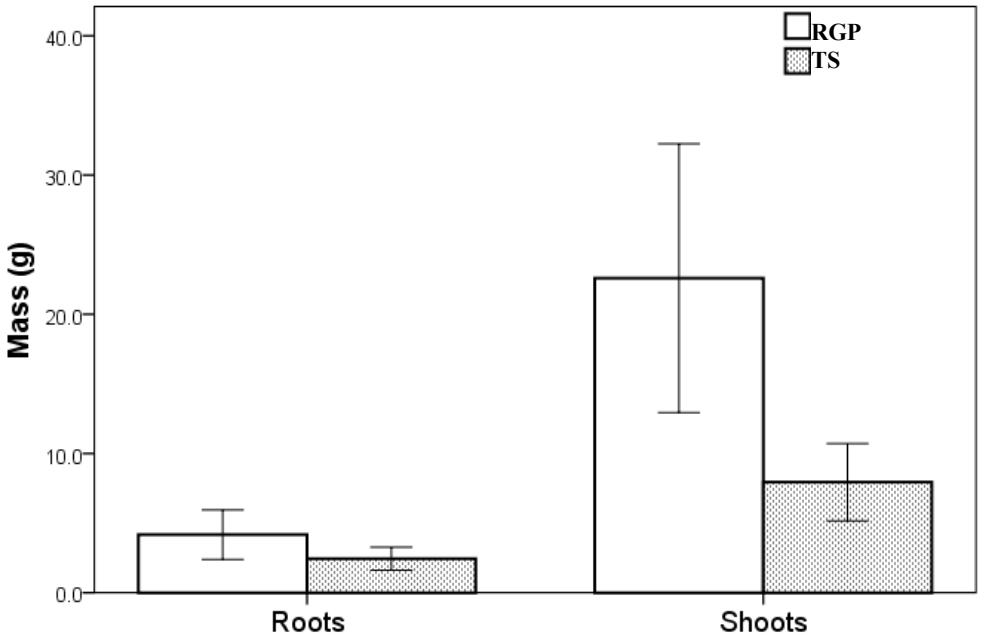

Figure 37. Mean mass (in grams) for roots and shoots of individual *S. scoparium* plants collected on the restored gravel pit (RGP) and naturally recovering target site (TS). Error bars represent +/-1 S.E.M.

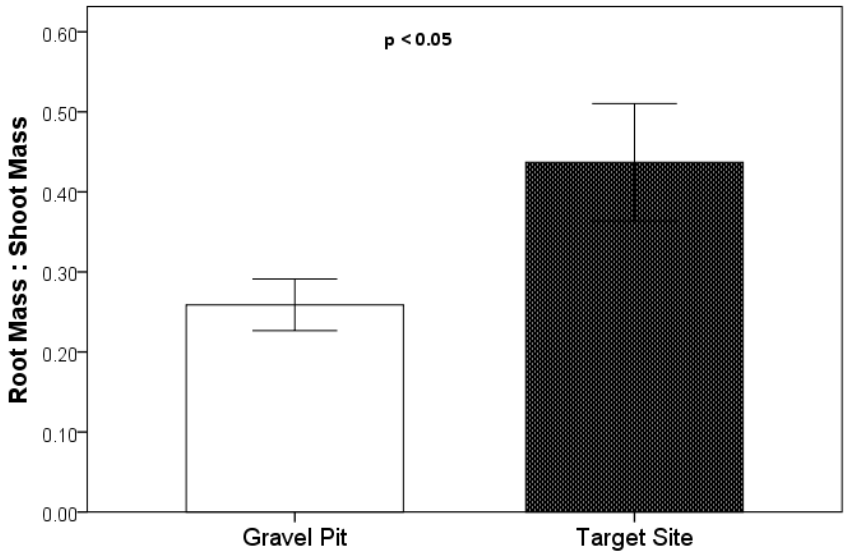

Figure 38. Mean ratio of root mass to shoot mass for individual *S. scoparium* plants collected on the restored gravel pit and target site. Error bars represent +/-1 S.E.M.

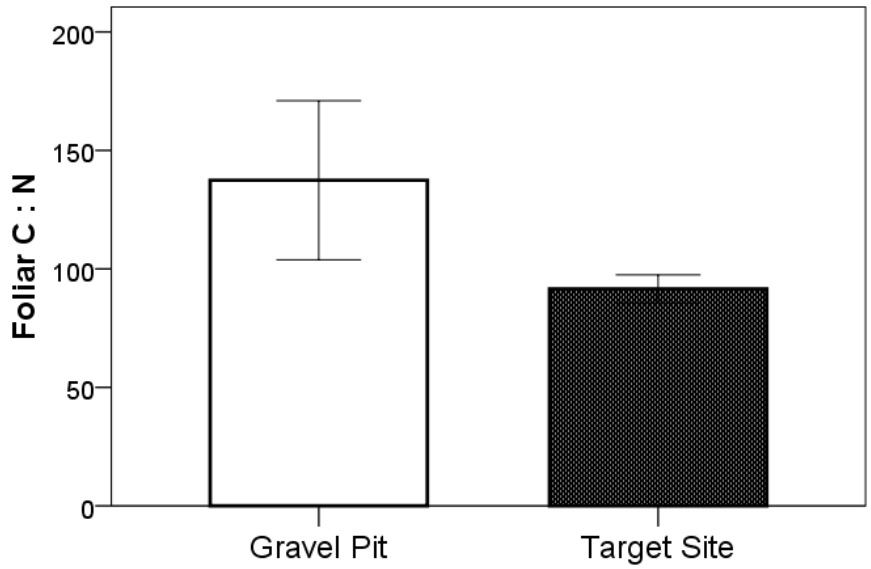

Figure 39. Mean foliar carbon to nitrogen ratio for *S. scoparium* plants collected on the restored gravel pit and target site. Error bars represent +/-1 S.E.M.

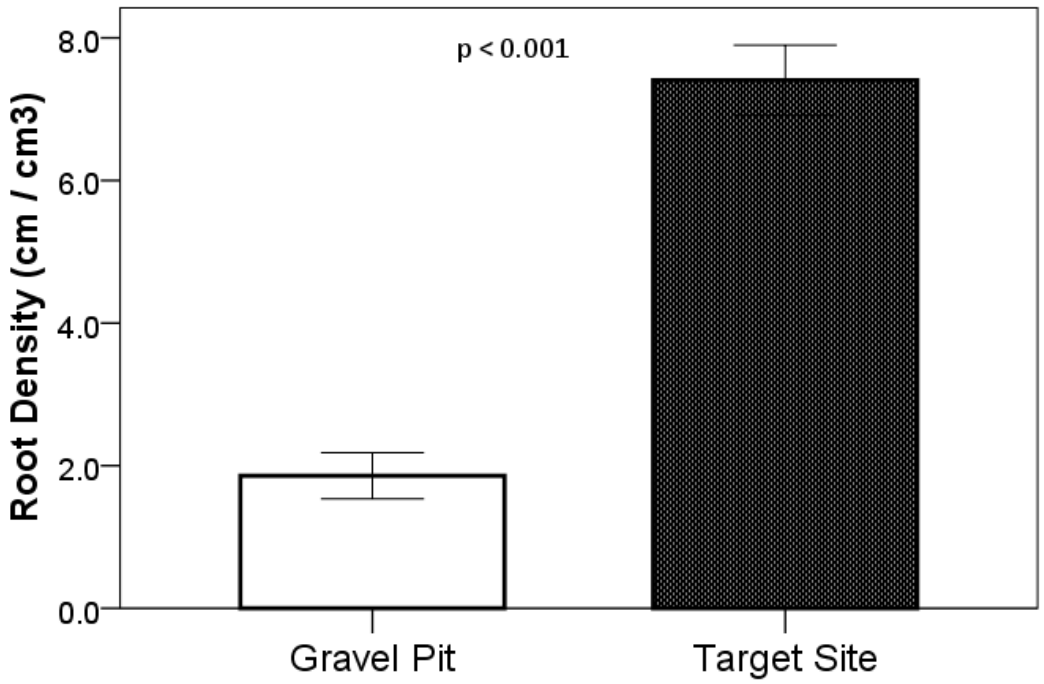

Figure 40. Mean root length density (cm /cm^3) obtained in soil cores collected from the restored gravel pit and target site. Error bars represent +/- 1 S.E.M.

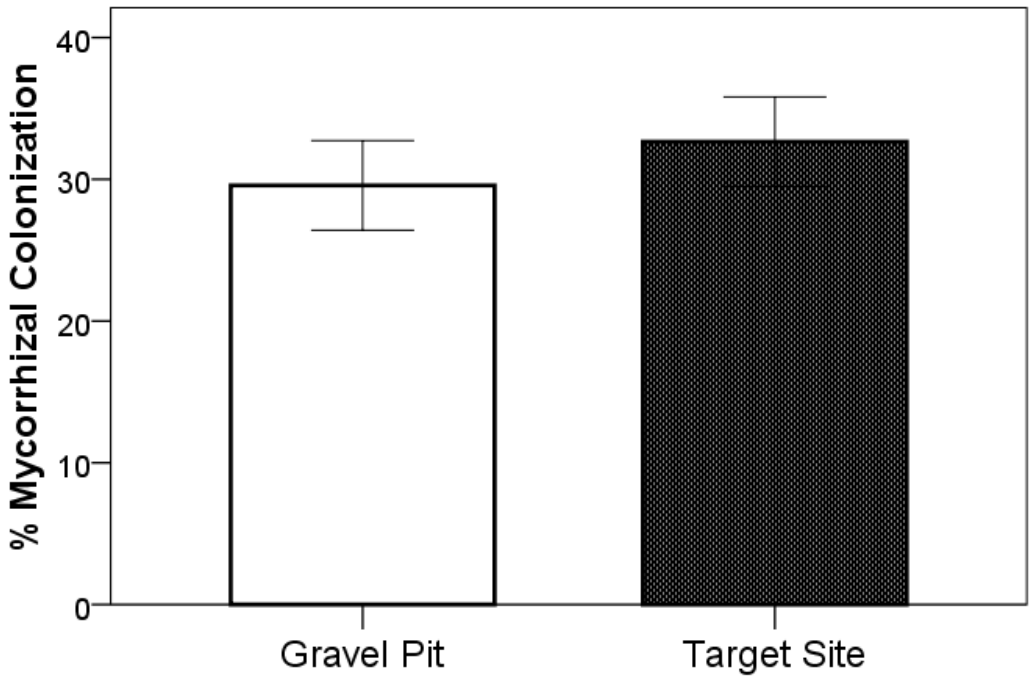

Figure 41. Mean root length colonized by mycorrhizae (%) on collected root samples of individual *S. scoparium* plants on the restored gravel pit and naturally recovering target site. Error bars represent +/- 1 S.E.M.

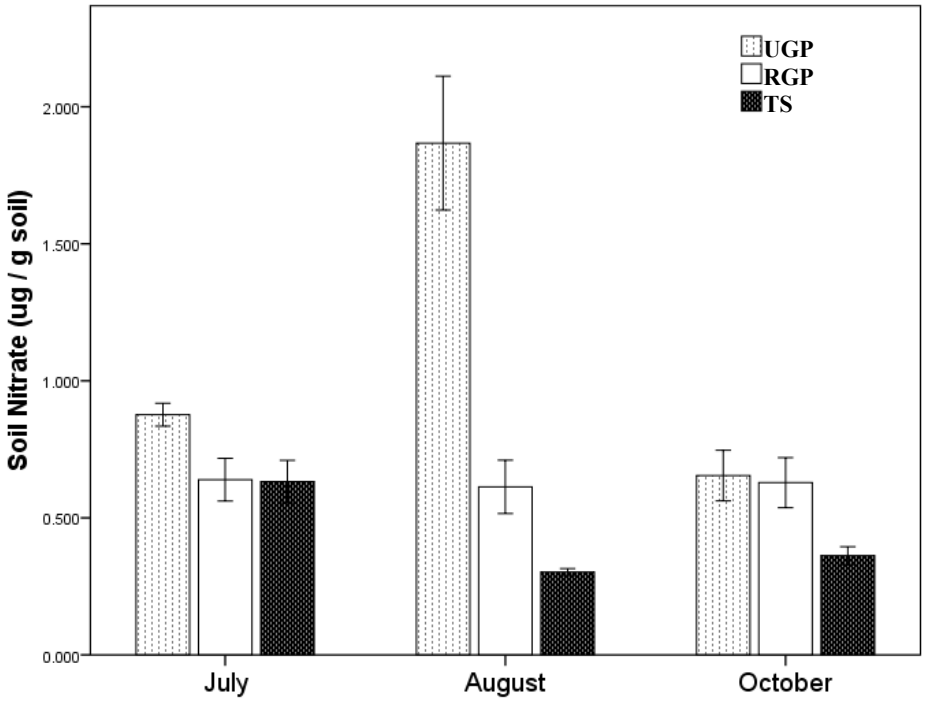

Figure 42. Mean soil nitrate (NO_3^-) in ug/g soil for each site for each sampling period. Error bars represent +/- 1 S.E.M. Mean differences among sites and months are reported in Table 22.

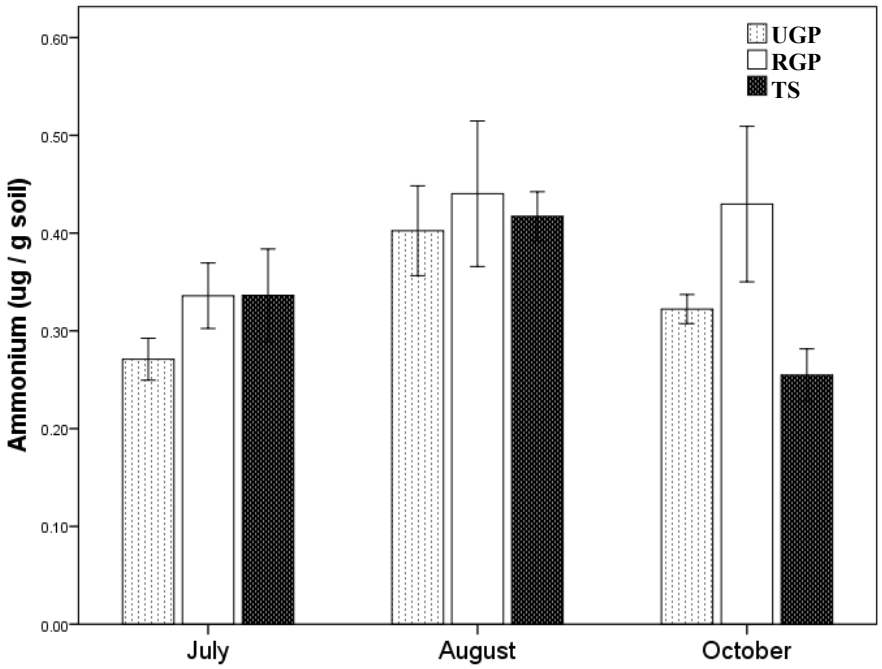

Figure 43. Mean soil ammonium (NH^{+}_{4}) in ug/g soil for each site for each sampling period. Error bars represent +/- 1 S.E. of the mean. Mean differences among sites and months are reported in Table 23.

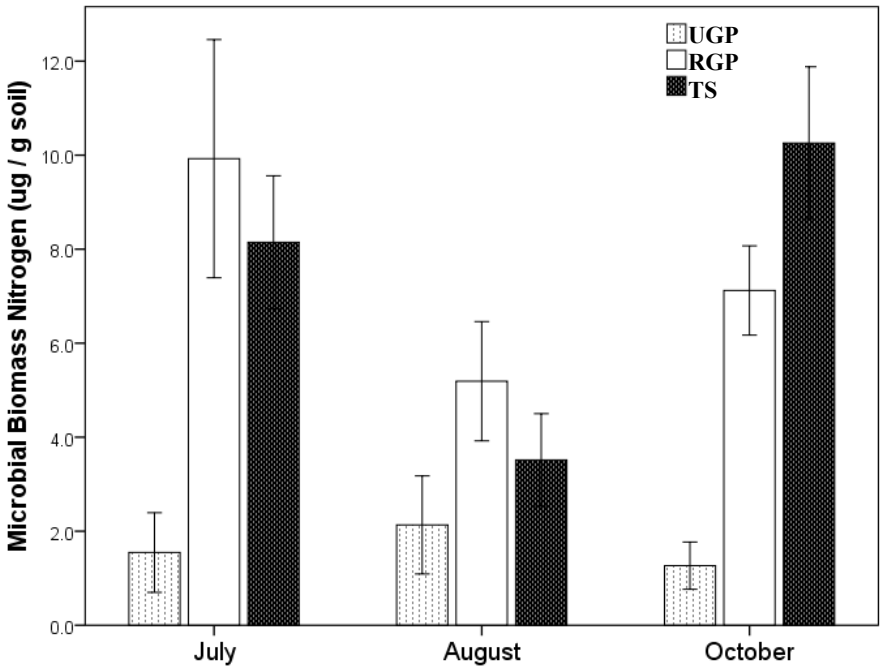

Figure 44. Mean microbial biomass nitrogen (MBN) in ug/g soil for each site for each sampling period. Error bars represent +/- 1 S.E.M. Mean differences among sites and months are reported in Table 24.

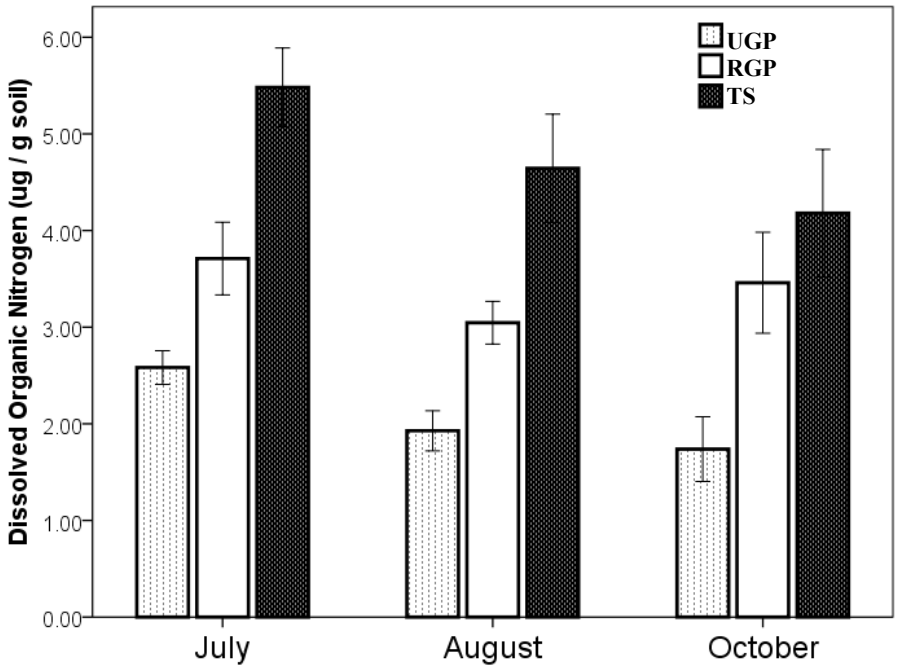

Figure 45. Mean dissolved organic nitrogen (DON) in ug/g soil for each site for each sampling period. Error bars represent +/- 1 S.E.M. Mean differences among sites and months are reported in Table 25.

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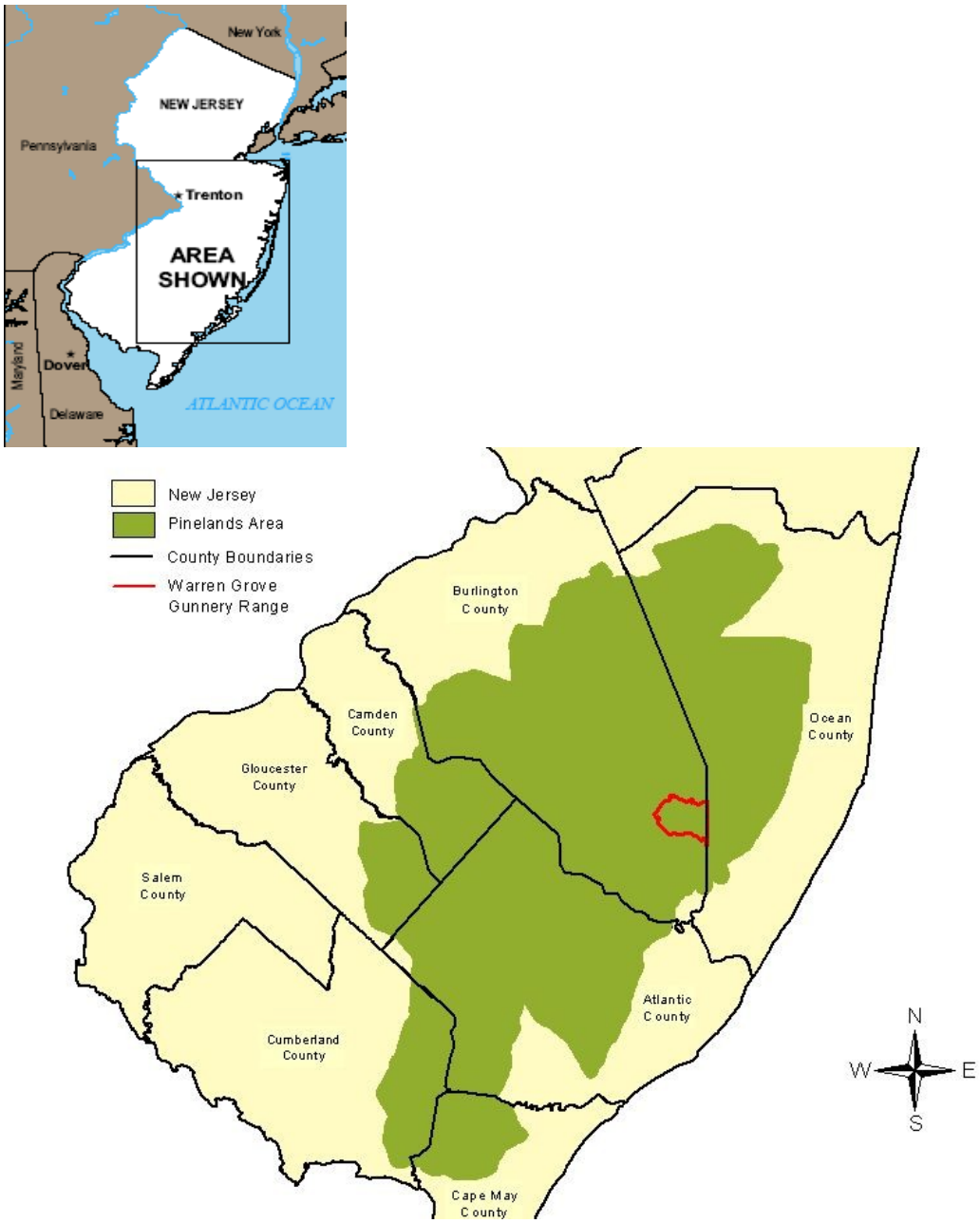
APPENDIX 1: Location of the Warren Grove Gunnery Range in south central New Jersey U.S.A.



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